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REVIEW

Impacts and prognosis of natural resource development on aquatic biodiversity in Canada's boreal zone[1]

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Abstract: Conservation efforts to sustain water resources and aquatic biodiversity in boreal watersheds will require reliable information on the recent status of various indicator species and an improved understanding of the risks to aquatic biodiversity posed by resource development activities. We reviewed the recent state of knowledge on the responses of aquatic biodiversity to forest management, pulp and paper mill effluents, hydroelectric impoundments, mining of minerals and metals, oil sands extractions, and peat mining and offer a prognosis for aquatic biodiversity under each of these environmental stressors. Despite the prevalence of natural resource development in Canada's largest forest ecosystem, there was a limited amount of published literature on the effects of many of the disturbance types on various indicators of aquatic biodiversity, making it difficult to produce a current and reliable status assessment. Across most of the boreal zone, there is a lack of coordinated, consistent data collection for many of the bioindicators and disturbance types discussed in this review. Forecasting the future state of aquatic biodiversity across the boreal zone is challenged by increasing natural resource development and its interactions with other stressors, especially climate change. The cumulative effects of multiple stressors coupled with resource development activities in boreal watersheds remain largely unknown. More importantly, the ecological thresholds for these cumulative effects (that is, the point at which aquatic ecosystems and their biodiversity cannot recover to a desired state within a reasonable time frame) are also unknown and remain gaps in our knowledge. The recent literature identifies a number of risks to aquatic biodiversity at local (tens of square kilometres) to regional (hundreds of square kilometres) scales associated with natural resource development. There are indications that many of these risks can be minimized by "greener" technologies for resource development and reclamation, practical conservation planning and regulation, and increased stewardship in watershed management, although the effectiveness of many of these measures cannot yet be assessed from the published literature.

Key words: boreal, watershed, resource development, impact, prognosis.

Résumé : Les efforts de conservation pour maintenir les ressources hydriques et la biodiversité aquatique dans les bassins versants boréaux nécessitera de l'information fiable sur l'état des diverses espèces indicatrices et une meilleure compréhension des risques pour la biodiversité aquatique générés par les activités de développement des ressources. Les auteurs ont passé en revue l'état des connaissances actuelles sur les réactions de la biodiversité aquatique à l'aménagement forestier, aux effluents des papeteries, aux barrages hydroélectriques, à l'extraction des minéraux et des métaux, à l'extraction des sables bitumineux ainsi que de la tourbe et ils offrent un pronostic pour la biodiversité aquatique soumise à ces différents stress environnementaux. En dépit de la prévalence du développement des ressources naturelles dans les écosystèmes les plus vastes du Canada, on observe une quantité limitée de littérature publiée sur les effets de plusieurs des types de perturbation sur les divers indicateurs de biodiversité aquatique, rendant difficile la production d'une évaluation fiable de la situation actuelle. Sur la plus grande partie de la zone boréale, il y a un manque de récolte de données coordonnées et consistantes sur plusieurs des bio-indicateurs et types de perturbation discutés dans cette revue. La prédiction de l'état futur de la biodiversité aquatique sur l'ensemble de la zone boréale est remise en question par l'augmentation du développement des ressources et ses interactions avec d'autres agents stressants, surtout le changement climatique. Les effets cumulatifs des multiples agents stressants couplés avec les activités de développement des ressources dans les bassins versants boréaux demeurent largement inconnus. Encore plus important, les seuils écologiques pour ces effets cumulatifs (soit le point à partir duquel les écosystèmes aquatiques et leur biodiversité ne peuvent pas recouvrer un état attendu, dans un espace de temps raisonnable) demeurent également inconnus et constituent toujours une brèche dans notre connaissance. La littérature récente identifie un nombre de risques pour la diversité aquatique aux échelles locales (dizaines de km2) à régionales (centaines de km2) associés au développement des ressources naturelles. Il y a des indications à l'effet que plusieurs de ces risques pourraient être minimisés par des technologies 'plus vertes' pour le développement des ressources et leur restauration, la planification et la réglementation des pratiques de conservation et l'augmentation de la gérance dans l'aménagement des basins versants, bien que l'efficacité de plusieurs de ces mesures ne peut toujours pas être évaluée à partir de la littérature publiée. [Traduit par la Rédaction]

Mots-clés : boréale, bassin versant, développement des ressources, impact, pronostique.

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1. Introduction

The myriad of water bodies across Canada's boreal zone, including lakes, ponds, rivers, streams, and wetlands, support an array of aquatic biodiversity. There are few published studies that attempt to quantify total aquatic species richness, but based on archived records from boreal and sub-boreal regions (S. Laframboise. Personal communication, 2012), it is estimated that Canada's boreal zone supports about 200 fish species, 21 species of amphibians, 26 species of reptiles (Canadian Amphibian and Reptile Conservation Network 2010), and more than 250 species of wetland plants (Wetlands Alberta 2011). Most of these are endemic, although overall non-native species introductions are increasing across the boreal zone (Langor et al. Manuscript in preparation). Estimates of aquatic species richness are not available for lower order organisms, but anecdotal evidence from numerous studies and surveys indicates that water bodies of the boreal zone support thousands of species among aquatic invertebrate, zooplankton, phytoplankton, rotifer, and microbial assemblages (e.g., Dodson et al. 2000; Vinebrook et al. 2003; Spitzer and Danks 2006). Biodiversity in water bodies of the boreal zone is low in comparison with that in similar habitat types of temperate or tropical zones because of the relatively harsh climatic conditions and nutrient-poor waters. Nevertheless, aquatic biodiversity in the boreal zone serves comparable critical functions to its temperate counterparts in the maintenance of healthy aquatic ecosystems and their associated ecosystem services, including production of clean water, carbon storage, nutrient uptake, biogeochemical cycling, food web subsidies, and flood control (Woodward 2009; Wells et al. 2010).

In forested landscapes, aquatic systems and their biodiversity are ecologically linked to their surrounding terrestrial watersheds (Hynes 1975; Richardson and Danehy 2007). Therefore, the ecological integrity (structure and function) of aquatic ecosystems could be directly threatened by disturbances to the water bodies themselves or indirectly threatened through disturbances to boreal forest watersheds. Schindler (1998*a*) pointed out that the relatively simple biological diversity in aquatic systems of the boreal forest renders those aquatic communities particularly vulnerable to disturbances owing to reduced redundancy among functionally similar species. A particular disturbance that reduces or eliminates a specific taxon or functional group could compromise critical community or ecosystem functions because there are few or no similar species available to adapt to the changed conditions and fill the compromised niche (e.g., Vinebrook et al. 2003; Kreutzweiser et al. 2004).

The North American boreal zone is under increasing industrial exploitation pressure in the face of looming climate change impacts (Henry 2005; Schindler and Lee 2010; Wells et al. 2010) and this could potentially put aquatic ecosystems and their biodiversity at risk. Meaningful impact mitigation strategies and conservation plans for sustaining aquatic biodiversity in boreal watersheds will require reliable information on the recent and current status of various indicator species and an improved understanding of the risks to aquatic biodiversity posed by resource development activities.

The purpose of this review is to synthesize our state of knowledge on the status of and risks to aquatic biodiversity in the boreal zone of North America. This paper is one of a series in which risks and threats to Canada's boreal zone are examined (Brandt et al. 2013). In this review we include fish, amphibians (larval aquatic and adult shoreline stages), macroinvertebrates, zooplankton, phytoplankton, periphyton, and macrophytes (plants) as indicators of aquatic biodiversity. It is unfeasible to study all aquatic species, and consequently research and monitoring typically default to selected indicator species or groups. For each of these taxa, we review the impacts of natural resource development and use, based on available information. Prevalent disturbances in the boreal zone that we included are forest management (harvesting, roads, pest control), hydroelectric impoundments, pulp and paper effluents, mining effluents and tailings, oil and gas exploration and development, and peat mining.

The recent and relevant literature were searched primarily using Google™, Google Scholar™, and Scopus® (from ScienceDirect®). Although the focus of the review was on the Canadian boreal zone, information was occasionally drawn from studies in other forest regions when specific information from the boreal zone was not available or when the other studies elucidated risks applicable to boreal ecosystems. Emphasis was on published literature reporting recent impacts, status, or trends (generally, all publications on the topic of interest from within the last 15 years were included in our review), but select older studies were included to provide context or to augment the findings when recent publications were few, and some unpublished reports or nonrefereed publications were included when the published literature on a topic was scarce. The reviewed literature is listed and summarized in tables and the response measurements were directional (increases or decreases) and categorized as large, moderate, or little or none. This rating of response measurements was a subjective process. We had no formal, quantitative thresholds for assigning categories but rather, the rating was intended to provide a relative summary-metric of response to the disturbance agent. Statistically significant effects of months to years or more than a generation in length were usually classified as large; those that were shorter term but significant and (or) detectable and obvious were usually classified as moderate; and those that were nonsignificant, not measurably different from controls or references, and (or) were short-term (in the order of days to weeks) were usually classified as little or none.

We recognize that important disturbances not directly related to resource development also affect and potentially threaten aquatic ecosystems and their biodiversity in Canada's boreal zone and that they may have compounding and (or) confounding influences on resource development impacts. These disturbances include wildfire, windthrow, insect damage, atmospheric pollution, acidification, and climate change. Some of these disturbance agents (e.g., wildfire, windthrow) are considered natural in that the recovery states from such disturbances serve as reclamation or conservation targets for resource management and healthy boreal forests. Many of these disturbances have been reviewed or considered in previous publications and are therefore not explicitly dealt with here. Previous publications with information on boreal ecosystems include reports on status and trends (Urquizo et al. 2000; FPTC 2010), conservation (NRTEE 2010; Schindler and Lee 2010; Wells et al. 2010), impacts of fire (St-Onge and Magnan 2000; Patoine et al. 2002*a*; Bisson et al. 2003; Pilliod et al. 2003; Tonn et al. 2004; Neary et al. 2008), and impacts of climate change (Magnuson et al. 1997; Schindler 1998*b*; Lake et al. 2000; Schindler 2001; Arnott et al. 2003; Donahue et al. 2003; Baulch et al. 2005; Chu et al. 2005; Keller 2007; Corcoran et al. 2009; Heino et al. 2009; Xenopoulos et al. 2009; Clarke et al. 2010; Alahuhta et al. 2011; Shimoda et al. 2011).

2. Risks to aquatic biodiversity

2.1. Forest management

Because of the strong land–water linkages in forest watersheds, management activities that change the forest composition or structure (Macdonald et al. 2010), the soil conditions, or biogeochemical cycling (Kreutzweiser et al. 2008*a*; Maynard et al. In press) or that alter the flow and quality of water entering aquatic systems (Buttle et al. 2000, 2005, 2009; Webster et al. Manuscript in preparation) can have impacts on the aquatic biota inhabiting those water bodies. Numerous studies (Webster et al. 1992; Pinel-Alloul et al. 2002; Prepas et al. 2003; Fortino et al. 2004; Steedman et al. 2004; Browne 2007; Richardson 2008) have reviewed the responses of aquatic organisms to watershed disturbances from forest management over the past few decades, many in nonboreal regions. In general they show that tree removal, the ground disturbance associated with such removal, the resultant changes to nutrient cycling and exports, and logging road construction are the main aspects of forest harvesting that can affect aquatic organisms in receiving waters. These disturbances can reduce canopy cover and shading, increase water temperatures, increase fine sediment deposition, increase nutrient concentrations, reduce large wood and fine organic matter inputs, and restrict fish movement. All of these have implications for the survival, production, and diversity of aquatic communities.

The reported severity and duration of impacts are highly variable among studies; they are often site-specific and usually related to the aerial extent of watershed disturbed and the proximity of harvested areas to shorelines. When harvesting is intense and near shorelines, impacts on aquatic communities can be long term, measurable for up to two decades or more (e.g., Ely and Wallace 2010). However, various best management practices to mitigate the impacts of forest harvesting on aquatic ecosystems, including the application of riparian (shoreline) buffers or reserves, are now widely implemented across North America (Lee et al. 2004; Vowell and Frydenborg 2004; Schilling 2009) and these have been largely effective in reducing many adverse effects on most aquatic organisms and their habitats (Broadmeadow and Nisbet 2004; Hickey and Doran 2004). In the Canadian boreal zone, mitigation measures and regulations to reduce impacts on aquatic ecosystems are improving (e.g., Creed et al. 2008) and increasingly being applied across various jurisdictions. Here we review recent studies from the boreal zone on the impacts of two main forest management activities on aquatic biodiversity: forest harvesting (see Fig. 3 in Brandt et al. (2013) for the area of commercial forest across the boreal zone) and pesticide applications, with emphasis on the effectiveness of impact mitigation methods.

2.1.1. Harvesting and roads

2.1.1.1. Fish

Fish populations can be adversely affected by forest harvesting, although impacts on fish and fish habitats depend on the fish species present, the type of forestry operations, and the potential secondary impacts arising from the effects of harvesting on water quality, temperatures, lower trophic levels, and food webs (Carignan and Steedman 2000). In comparison to earlier forest harvesting practices, which often had dramatic effects on fish populations (Hicks et al. 1991), most current or recent forest harvesting operations in the Canadian boreal zone have subtle or infrequent adverse effects on fish. Among the 16 studies or reviews we examined, 11 reported few or no measurable adverse impacts, and two of those reported positive effects (increased fish survival and growth) (Table 1). Most negative effects (reductions) on fish growth, survival, or habitat quality resulted from increased fine sediment loads in streams (Anderson 1996; Ripley et al. 2005; Browne 2007; Scrimgeour et al. 2008) or problematic culvert installations (Browne 2007; Park et al. 2008). Harvesting and road construction activities that produce fine sediments exceeding tolerance thresholds in streams or shorelines can negatively affect fish populations by reducing water and oxygen exchange in spawning gravel and by restricting intergravel movement of alevins, thereby reducing the survival of eggs and juvenile fish (Scrivener and Brownlee 1989; Curry and MacNeil 2004). In most cases, road construction and maintenance pose a far greater risk of fine sediment inputs to water bodies than the harvesting activities themselves (Croke and Hairsine 2006).

Road construction during forest harvesting can also adversely affect fish populations. Improper culvert installation, including perched, undersized, poorly oriented, clogged, or unstable culverts under forest roads can restrict fish movement, cause scouring from high-velocity restricted flow, increase fine sediment loading, and disrupt resting or nursery pools (Park et al. 2008). Roads, trails, and other soil compaction can disrupt surface and subsurface water flow and alter patterns in upwelling groundwater at critical fish spawning and juvenile habitat sites (Curry et al. 2002), although the effects of such disruptions on actual fish survival and reproduction are not clear. Forest roads can also pose a risk of adverse effects on fish populations by increasing access to otherwise pristine water bodies that can result in overexploitation of game fish (Gunn and Sein 2000) or by increasing introductions of non-native fish that threaten native fish diversity (Vander Zanden et al. 2004; Kaufman et al. 2009). Given that these studies of forestry impacts in boreal water bodies have identified the primary risks to fish and fish habitat, it appears that most adverse effects can be minimized by careful logging practices, proper road placement and construction, and improved culvert installation and water-crossing methods. Management of the excessive exploitation is a more difficult issue and may include strategic road placement, effective road decommissioning and controls on public access.

2.1.1.2. Amphibians

Amphibian populations may be at risk from forest harvesting primarily through the loss of or reduction in critical habitat. Canopy removal and hydrological disruptions from harvesting and road construction can change adult and larval amphibian habitats and pose a risk of harm to populations. These risks are cause for concern because a global decline in amphibians is recognized as a signal of deteriorating environmental conditions worldwide (Houlahan et al. 2000), and boreal ecosystems are known to be critical amphibian habitats (Wells et al. 2010). Although studies from nonboreal forest regions have shown that clear-cut harvesting can induce habitat losses and have significant short-term effects (often reduced abundance) on amphibians (DeMaynadier and Hunter 2000; Marczak et al. 2010; Popescu and Hunter 2011, Freidenfelds et al. 2011), studies on the effects of forest harvesting in the boreal zone indicated that current or recent forest management practices pose little risk of significant adverse effects on amphibians (Table 1). Of the seven studies or reviews that we found, five reported little or no measurable impacts of forest harvesting. Hamilton et al. (1998) reviewed the status of the Canadian toad (*Bufo hemiophyrs*) in Alberta and implicated forest harvesting in the decline of hibernation sites, but conceded that specific causes of decline were difficult to determine and required further study. More recently, Constible et al. (2010) found that the movement patterns of the same species in boreal forests and wetlands put them at risk from forest harvesting, when harvesting activities encroached on specific habitats. Given the increasing international attention being paid to global amphibian declines, the scarcity of empirical studies on amphibians in the boreal zone is of concern. A rigorous assessment of the impacts of current forest harvesting practices on amphibians in Canadian boreal water bodies and their surrounding habitats will require additional study. See Venier et al. (Manuscript in preparation.) for further information on upland amphibian population trends.

2.1.1.3. Macroinvertebrates

Aquatic macroinvertebrate communities are widely recognized as sensitive, effective, and ecologically relevant bioindicators of watershed and aquatic ecosystem disturbances (Rosenberg and Resh 1993). Numerous studies have assessed the effects of forest harvesting on aquatic macroinvertebrates (Fortino et al. 2004; Richardson 2008), but comparatively few have been conducted in boreal forest watersheds (Table 1). From other forest regions it is clear that intense forest harvesting, especially clear-cutting to shorelines, can have large and long-lasting effects on aquatic macroinvertebrate communities (Campbell and Doeg 1989; Webster et al. 1992; Stout et al. 1993; Growns and Davis 1994; Wood and Armitage 1997; Stone and Wallace 1998). The effects of such

Table 1. Impacts of forest management practices on aquatic biodiversity.

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
Fish	Forest harvesting and roads					
	Riparian harvesting	Canada	Review	Fish survival; growth	- or +; + (species dependent)	Browne 2007
	Riparian harvesting	British Columbia	3 years	Sockeye salmon emergence; emigration	- or 0; - or 0 (dependent on climate)	Macdonald et al. 1998
	Watershed harvesting (8%–73%)	Quebec	Review, 3 years	Fish populations	0	Pinel-Alloul et al. 2002
	Watershed harvesting (20% upland with 9% of stream length), no buffer	Newfoundland	3 years (2 pre, 1 post)	Brook trout populations (inferred from water temperatures)	0	Scruton et al. 1998
	Two intensive cuts, one moderate cut with buffer	Ontario	10 years (5 pre, 5 post)	Fish communities; abundance; size	0; 0; 0	Steedman 2003
	Clear-cut (33%) with buffer or clear-cut upland (60%–70%) and shoreline (40%–60%)	Ontario	8 years	Lake trout habitat volume	0	Steedman and Kushneriuk 2000
	Clear-cut (8.5%–73.2%)	Quebec	3 years	Abundance; growth; size structure	0; 0; - (fewer small fish)	St-Onge and Magnan 2000
	Watershed harvesting (15%–27.3%), buffer ≥ 100 m	Alberta	3 years	Lentic fish assemblage	0	Tonn et al. 2003
	Riparian clear-cutting (20%) and road construction	Newfoundland	2 years	Brook trout movement; habitat selection	0; -	McCarthy et al. 1998
	Road networks and harvesting (10%–40%)	Alberta	7 years	Abundance of bull trout	–	Ripley et al. 2005
	Road density	Alberta	6 years	Structure and occurrence of fish assemblages	- or + (species dependent)	Scrimgeour et al. 2008
	Forestry roads, loss of habitat, increased angling	Ontario	9 years	Lake trout populations (owing to habitat loss); exploitation (angling)	0; –	Gunn and Sein 2000
	Road culverts	Canada	Review	Migration	–	Browne 2007
	Stream fragmentation owing to hanging culverts	Alberta	2 years	Inferred fish diversity	-	Park et al. 2008
	Increased sediment load	Canada (not all boreal)	Review	Growth; swimming; health; egg survival; habitat	-; -; -; -; -	Anderson 1996
	Increased sediment load	Canada	Review	Salmonid spawning; brook trout abundance	-; -	Browne 2007
	Pesticides					
	Insecticides and herbicides	Ontario	Review	Survival	0	Steedman and Morash 2001
	Herbicide glyphosate	British Columbia	2 weeks after spray	Survival; behaviour	0; 0	Hildebrand et al. 1982
	Herbicide triclopyr ester	Ontario	35 days after spray	Survival; behaviour	–; –	Kreutzweiser et al. 1995
Amphibians	Forest harvesting and roads					
	Harvesting with buffer	Alberta	Review	Breeding habitat; hibernacula habitat	0; -	Hamilton et al. 1998
	Modified clear-cut, 60 m buffers	Alberta	2 years	Anuran abundance	0	Constible et al. 2001
	Large-scale harvesting	Alberta	2 years	Movement of Canadian toad	- (loss of upland habitat)	Constible et al. 2010
	Clear-cut, buffers > 20 m	Alberta	3 years	Anuran abundance; species composition	0; 0	Hannon et al. 2002

Table 1 (*continued*).

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
	Clear-cut with buffers	Ontario	3 years	Salamander abundance; colour morphs	0; 0	Pearce and Venier 2009
	Clear-cut with buffers and herbicides	Ontario	2 years	Abundance	0	Thompson et al. 2008
	Clear-cut (21%)	Ontario	Model	Red-backed salamander abundance	0	Venier et al. 2007
	Pesticides					
	Herbicide triclopyr ester	Ontario	Laboratory, short term	Survival	–	Edginton et al. 2003
	Herbicide triclopyr ester	Ontario	5 months	Survival; behaviour; growth	-; -; 0	Wojtaszek et al. 2005
	Herbicide triclopyr ester	Vermont (nonboreal)	Laboratory, 8 days	Survival	–	Chen et al. 2008
	Herbicide glyphosate	Vermont (nonboreal)	Laboratory, 8 days	Survival	-	Chen et al. 2004
	Herbicide glyphosate	Ontario	Laboratory, short term	Survival	- or 0 (pH dependent, formulation dependent)	Edginton et al. 2004
	Herbicide glyphosate	Pennsylvania (nonboreal)	Several weeks	Survival; growth	-; –	Relyea 2004
	Herbicide glyphosate	Pennsylvania (nonboreal)	3 weeks	Survival	–	Relyea 2005
	Herbicide glyphosate	Ontario	1 year, biomonitoring	Survival	0	Thompson et al. 2004
	Herbicide glyphosate	Ontario	5 months	Survival; behaviour; growth	0; 0; 0 (at realistic concentrations)	Wojtaszek et al. 2004
	Herbicide glyphosate	New Brunswick	14 days	Juvenile survival; body condition; liver somatic index; bacterial infection	0; 0; - or 0; - or 0 (at realistic concentrations)	Edge et al. 2011
Macroinvertebrates	Forest harvesting and roads					
	Riparian clear-cutting	Ontario	1 year	Colonization abundances	0	France 1998
	Riparian clear-cutting (older and recently logged)	British Columbia (sub-boreal)	1 year	Biomass	++ (no impact in older logged streams)	Fuchs et al. 2003
	Partial riparian harvesting (10%, 21%, and 28% removal)	Ontario	7 years (2–3 pre, 3–4 post)	Leaf litter breakdown; macroinvertebrate community	0; 0	Kreutzweiser et al. 2010
	Partial harvesting without buffers	Ontario	5 years (2 pre, 3 post)	Macroinvertebrate abundance; community structure	0 or +; 0 or + (dependent on logging intensity)	Kreutzweiser et al. 2005
	Clear-cut, buffer 30–100 m	Ontario	1 year	Community richness; leaf-litter decomposition	–; -	Kreutzweiser et al. 2008*b*
	Clear-cut (33%–96%)	Alberta and Quebec	2 years	Density; richness; biomass	0; 0; + (species dependent)	Scrimgeour et al. 2000
	Proximity to shoreline, intensity, road density	Canada	Review	arthropod production; abundance; diversity	+; +; -	Richardson 2008
	Increased sedimentation from roads	Canada	Review	Communities	–	Browne 2007
	Pesticides					
	General overview of pesticides	Ontario	Review	Survival	0	Steedman and Morash 2001
	Insecticide Btk	British Columbia	7 days after spray	Density; community composition; emergence	0; 0; 0	Richardson and Perrin 1994

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Table 1 (*continued*).

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
	Insecticide Btk	Ontario	Laboratory (nonboreal)	Survival; feeding rates	0; 0	Kreutzweiser and Capell 1996
	Insecticide Btk	Ontario	9 days after spray	Survival; drift	0; 0	Kreutzweiser et al. 1992*a*
	Insecticide Btk	Ontario	4 months after spray	Drift; survival; abundance; growth	0; 0; 0; 0	Kreutzweiser et al. 1994*b*
	Insecticide Btk	Canada	Review	Communities; survival	0; 0	Perrin et al. 1995
	Insecticide tebufenozide	Laboratory (nonboreal)	48 hours after dose	Survival	0 (at realistic concentrations)	Song et al. 1997
	Insecticide tebufenozide	Ontario	Up to 12 days	Survival; drift; feeding rates	0; 0; 0	Kreutzweiser et al. 1994*c*
	Herbicide glyphosate	Laboratory (nonboreal)	Days	Survival	0 (at realistic concentrations)	Folmar et al. 1979
	Herbicide glyphosate	British Columbia	9 days after spray	Drift	0	Kreutzweiser et al. 1989
	Herbicide triclopyr ester	Ontario	2 days after spray	Drift; survival	0; 0	Kreutzweiser et al. 1992*b*
	Herbicide triclopyr ester	Ontario	8 days after spray	Survival; feeding rates	0; 0 (at field concentrations)	Kreutzweiser et al. 1998*a*
	Herbicide triclopyr ester	Ontario	Up to 3 months after spray	Drift; abundance	-; 0	Thompson et al. 1995
	Herbicides (both)	Canada	Review	Survival	0 (at realistic concentrations)	Roshon et al. 1999
Zooplankton	Forest harvesting and roads					
	Clear-cut (9%–72%), 20 m buffers	Quebec	3 years	Health inferred from methylmercury accumulation	-	Garcia et al. 2007
	Clear-cut (43%–73%)	Quebec	3 years	Richness; assemblages; biomass	0; 0; 0	Jalal et al. 2005
	Clear-cut (9%–73%), 20 m buffers	Eastern Canada	1–2 years	Crustacean size	0	Patoine et al. 2002*a*
	Clear-cut (9%–73%), 20 m buffers	Eastern Canada	1 year	Richness; assemblages	0; 0	Patoine et al. 2002*b*
	Clear-cut (7%–73%), 20 m buffers	Quebec	3 years	Total biomass	0 (species dependent decline short-term)	Patoine et al. 2000
	Watershed harvesting (8%–73%)	Quebec	Review, 3 years	Total biomass	0 or -(dependent on logging intensity)	Pinel-Alloul et al. 2002
	Watershed harvesting (0%–35%), buffer width 20–200 m	Alberta	4 years (2 pre, 2 post)	Abundance	-	Prepas et al. 2001
	Watershed harvesting (up to 70%)	Quebec, Alberta	Review	Cladoceran abundance; calanoid abundance	-; -	Prepas et al. 2003
	Watershed harvesting (40%–65%)	Quebec	1 year (two samples pre, two samples post)	Community structure	0	Winkler et al. 2009
	Pesticides					
	Insecticide tebufenozide	Laboratory (nonboreal)	48 hours after dose	Survival	0 (at realistic concentrations)	Song et al. 1997
	Insecticide tebufenozide	Ontario	Up to 1 year after spray	Abundance	– or 0 or + (species dependent)	Kreutzweiser and Thomas 1995
	Insecticide tebufenozide	Ontario	Several months	Abundance; community structure	0; 0	Kreutzweiser et al. 1998*b*
	Herbicide triclopyr ester	Ontario	Several months	Abundance, community structure	-; -	Wojtaszek 2004

Table 1 (*concluded*).

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
	Herbicide glyphosate	Laboratory (nonboreal)	Weeks	Survival; reproduction; development	–; –; –	Chen et al. 2004
	Herbicide glyphosate	Sprayed pond (nonboreal)	5 days after spray	Survival	0	Trumbo 2005
	Herbidicide glyphosate	Canada	Review	Survival	0 (at realistic concentrations)	Roshon et al. 1999
	Herbicide glyphosate	British Columbia	8 days after spray	Survival	0	Hildebrand et al. 1980
	Herbicide glyphosate	Ontario	Several months	Abundance; community structure	- or 0; - or 0	Wojtaszek 2004
Phytoplankton and periphyton	Forest harvesting and roads					
	Riparian clear-cutting (older and recently logged)	British Columbia	1 year	Chlorophyll *a* biomass	+ (recently logged)	Fuchs et al. 2003
	Clear-cut, 20 m buffer	Quebec	5 months	Diatom assemblage	+	Hausmann and Pienitz 2009
	Clear-cut: moderate (33% with buffer), extensive (60%–70% with shoreline harvesting 40%–60%)	Ontario	4 years	Phytoplankton communities	0	Knapp et al. 2003
	Clear-cut: moderate (45% with buffer ≥ 30 m), extensive (60%–70% with shoreline harvesting 40%–60%)	Ontario	10 years (5 pre, 5 post)	Phytoplankton biomass; richness; interannual variability in community structure	+; +; -	Nicholls et al. 2003
	Clear-cut, narrow or no buffer	Ontario	30 years	Chrysophytes	0	Paterson et al. 1998
	Watershed harvesting (15%–79%)	Quebec	2 years (BACI)	Periphyton communities	-	Desrosiers et al. 2006
	Clear-cut (43%–73%)	Quebec	3 years	Phytoplankton biomass	0 (species dependent)	Jalal et al. 2005
	Watershed harvesting (8%–73%)	Quebec	Review, 3 years	Phytoplankton biomass	+	Pinel-Alloul et al. 2002
	Watershed harvesting (9%–73%)	Quebec	3 years	Algal biomass; community structure	+; - (differential changes in structure)	Planas et al. 2000*a*
	Watershed harvesting (11%–96%)	Quebec	4 years	Periphyton biomass; phytoplankton biomass; richness	++; +; -	Planas et al. 2000*b*
	Watershed harvesting (0%–35%), buffer 20–200 m	Alberta	4 years (2 pre, 2 post)	Phytoplankton biomass	+ or - (species dependent)	Prepas et al. 2001
	Harvesting with buffers	Canada boreal	Review	Algal biomass	- or + (lake depth dependent)	Prepas et al. 2003
	Clear-cut	Michigan (nonboreal)	200 years (sediment cores)	Autotrophic community	- or 0 (species dependent)	Scully et al. 2000
	Watershed harvesting (40%–65%)	Quebec	1 year	Phytoplankton biomass	0	Winkler et al. 2009
	Pesticides					
	Herbicide triclopyr ester	Ontario	Several months	Abundance; community structure	+; 0 or +	Wojtaszek 2004
	Herbicide glyphosate	British Columbia	Weeks	Biomass	0	Sullivan et al. 1981
	Herbicide glyphosate	British Columbia	6 weeks after spray	Biomass; biovolume	+; +	Austin et al. 1991
	Herbicide glyphosate	Ontario	Several months	Abundance; community structure	+; 0 or +	Wojtaszek 2004

*The impact rating scale is as follows: –, large decrease; -, moderate decrease; 0, little or no measurable effect (where little is either a small or a brief change); +, moderate increase; and ++, large increase. End points are listed together, separated by a semicolon, for studies that examined multiple assessment end points. Btk, *Bacillus thuringiensis* var. *kurstaki*.

harvesting have generally resulted from increased fine sediment loading, reduced canopy cover, increased solar radiation and water temperatures, and changes to water runoff and quality. Most of these are scale dependent. It is generally accepted that watersheds with higher proportions of logged areas are more susceptible to logging impacts on water yield, quality, and other parameters (Steedman et al. 2004; Luke et al. 2007), although attempts at linking the scale of logging disturbance to aquatic ecosystem responses have produced variable results (Allan 2004; Martel et al. 2007; Richardson 2008). Buttle (2002) suggests that the impacts of forest harvesting are more likely to depend on the relative degree of hydrological connectivity (seasonally influenced) between upland areas and receiving waters than on the proportion of the watershed logged. Under more recent and improved forest management practices, adverse effects on aquatic macroinvertebrates and their habitats are often minimal (Carlson et al. 1990; Danehy et al. 2007; McCord et al. 2007; Hemstad et al. 2008; Gravelle et al. 2009) but some measurable changes in macroinvertebrate communities can still be detectable even after mitigation efforts, such as riparian buffers, are implemented (Kiffney et al. 2003; Martel et al. 2007).

Among the few boreal studies that we could find, the evidence indicates that most of the impacts of current or recent forest management practices on aquatic macroinvertebrates tend to be subtle and infrequent (Table 1). However, results varied and some appeared to be site specific for reasons that are not yet well understood. Reports from studies or reviews ranged from little or no measurable adverse effects on macroinvertebrate communities (France 1998; Scrimgeour et al. 2000; Fuchs et al. 2003; Kreutzweiser et al. 2005; Richardson 2008; Kreutzweiser et al. 2010) to significant declines in various community metrics after harvesting (Browne 2007; Kreutzweiser et al. 2008*b*; Richardson 2008). It appears that adverse effects on aquatic macroinvertebrates can be largely avoided by careful logging and other best management practices (Fortino et al. 2004; McCord et al. 2007; Ice et al. 2010). Nevertheless, the paucity of impact studies from the boreal zone, the site-specific and sometimes conflicting results, the generally short-term nature of these studies, and their tendency to focus on low-order watersheds indicate that spatially cumulative effects, temporally cumulative effects, and the hydro-ecological basis for conflicting results continue to be research gaps.

2.1.1.4. Zooplankton

Smaller, free-floating aquatic invertebrates (zooplankton) are also effective bioindicators of the health of aquatic ecosystems, especially standing water bodies. Forest harvesting primarily affects zooplankton in lakes and ponds through secondary effects of changes to water quality via altered nutrient concentrations and fluxes from watersheds and the resulting changes to food resources (Gregory et al. 1987). Water quality impacts, in turn, are highly dependent on watershed geomorphology, water flow paths, soil types, biogeochemical processing, groundwater and surface water residence times, postdisturbance weather patterns, and the extent and type of forest harvesting (Kreutzweiser et al. 2008*a*; Webster et al. Manuscript in preparation). Therefore, the risk of impacts on zooplankton is likely to be higher where any or all of these features or processes have been altered by forest harvesting operations. Shallow lakes with strong hydrological connections to the surrounding catchment and with short water renewal times (large watershed to volume ratios) will be at higher risk than deep lakes with long water renewal times (Rask et al. 1998; Prepas et al. 2003).

Most studies of the impacts of forest harvesting on zooplankton in boreal lakes in the past decade reported few, transient, or no measurable effects (Table 1). In general, significant changes to zooplankton communities were only detected in boreal lakes with large watershed to volume ratios (usually >4) and with more than 40% of the watersheds harvested (Pinel-Alloul et al. 2002; Prepas et al. 2003). Significant decreases in abundance of zooplankton (cladocerans and calanoids) after harvesting around lakes on the Boreal Plain were linked to decreased edible algae and increased inedible blue–green algae because of increased phosphorus concentrations (Prepas et al. 2001, 2003).

2.1.1.5. Phytoplankton and periphyton

Free-floating (phytoplankton) and attached (periphyton) algae are important primary producers in aquatic ecosystems. Primary production in open standing waters of boreal forests, particularly on the Boreal Shield, is often nutrient, especially phosphorus, limited (Dillon et al. 1988). Primary production in headwater streams with closed canopies is usually light limited (Vannote et al. 1980). Therefore, forest harvesting operations that significantly increase nutrient fluxes or light levels to water bodies are likely to increase primary production and potentially change algal community structure (Prepas et al. 2001; Kiffney et al. 2004).

Most studies reporting phytoplankton or periphyton responses to boreal watershed disturbances from forest harvesting were conducted in lakes. Among the 14 studies or reviews we examined, only four reported no measurable effects while most of the remaining ones detected measurable increases in algal community attributes (Table 1). Increased periphyton or phytoplankton biomass in response to logging-induced nutrient enrichment was considered a eutrophication effect. This was sometimes accompanied by a change in community structure, including shifts toward nonedible algal forms (Prepas et al. 2001). The studies indicated that algal responses to watershed disturbances tended to be more pronounced and longer lasting in Boreal Plain lakes than on the Boreal Shield, owing to higher movement of sediment-borne phosphorus runoff from harvested watersheds on the Plain (Pinel-Alloul et al. 2002; Prepas et al. 2003).

2.1.2. Pesticide applications

Historically, many forest pesticides have had measurable and often severe toxic effects on nontarget aquatic organisms (e.g., Kerswill and Edwards 1967; Symons 1977), so forest pest managers and regulators have increasingly relied over the past couple of decades on reduced-risk pesticides in Canada (Thompson and Kreutzweiser 2007). Here we consider only those pesticides that are currently registered and being used for forest management. The vast majority of forest pesticides being applied across Canada are either for insect control (insecticides) or competing vegetation control (herbicides) and this review is restricted to those classes.

Forest pesticide use overall is in decline. The total area of forest sprayed with insecticides (including nonboreal forests) in the early 1990s was about 370 000 ha (down from over 1 million ha in the 1970s). The total area sprayed in 2009 was 132 000 ha and in 2010 was 160 000 ha (National Forest Database 2013). In recent years, the biological insecticide *Bacillus thuringiensis* var. *kurstaki* (Btk) has constituted over 90% of the insecticide used in forest pest management (Thompson 2011). The molt-accelerating insecticide tebufenozide represented most of the remaining 10%. Forest herbicides were sprayed on about 170 000 ha in 1992, 110 000 ha in 2009, and 120 000 ha in 2010. Glyphosate is by far the most commonly used forest herbicide in Canada (constituting about 95% of total herbicide use) with triclopyr ester being the next most common (a little less than 5%; Thompson 2011). Although the herbicide 2,4-D was used on several thousand hectares per year in Canadian forests as recently as the early 2000s, its use has dwindled to less than a few hundred hectares in recent years (National Forest Database 2013). None of the pesticides currently used in Canadian forestry are considered persistent or likely to bioaccumulate, thus reducing the overall risks to aquatic organisms.

2.1.2.1. Fish

Neither of the forest insecticides currently being used in Canada's boreal forest (Btk and tebufenozide) pose a significant risk of direct harm to fish at expected environmental concentrations (Surgeoner and Farkas 1990; Wing and Aller 1990; Steedman and Morash 2001). The herbicide glyphosate, and (or) its forestry formulations containing the polyethoxylated amine surfactant (POEA), can be toxic to fish but many studies have demonstrated that expected field concentrations of glyphosate are unlikely to cause direct harmful effects on fish (Hildebrand et al. 1982; Solomon and Thompson 2003). By comparison, the herbicide triclopyr ester is highly toxic to fish in laboratory bioassays (Johansen and Geen 1990; Kreutzweiser et al. 1994*a*) and at least one field experiment demonstrated significant adverse effects on fish survival and behaviour at realistic field concentrations when triclopyr ester persisted for at least 24 h (Kreutzweiser et al. 1995). However, given that triclopyr ester represents a small fraction of the total herbicide used in Canadian forests and that regulations prohibit herbicide use over or near open water, risks to fish in the boreal zone from triclopyr applications will be small and localized.

2.1.2.2. Amphibians

Based on laboratory findings, the insecticides Btk and tebufenozide are not considered to be toxic to amphibians at realistic concentrations owing to their insect-specific mode of action (Thompson and Kreutzweiser 2007) but no field experiments to assess the toxicity of these insecticides to amphibians in forest-use settings appear to have been conducted. The herbicide triclopyr ester poses a risk of significant harm to larval stages of amphibians at expected environmental concentrations for forestry (Edginton et al. 2003; Wojtaszek et al. 2005; Chen et al. 2008) but as indicated earlier, the use of this herbicide in Canadian forestry is minimal and regulations prohibit applications of triclopyr ester on or near open water.

Most research into the toxicity of forest pesticides to amphibians is focused on the herbicide glyphosate because of its prevalent use and because the potential harm to amphibians remains a contentious issue. As with the potential toxicity to fish, some glyphosate formulations can be toxic to amphibians, particularly larval stages, even at relatively low concentrations. Some laboratory or microcosm studies indicate that realistic concentrations of glyphosate can have significant harmful effects on survival, growth, metamorphosis, development, or reproduction (Chen et al. 2004; Edginton et al. 2004; Relyea 2004, 2005). However, actual field studies in boreal wetlands have demonstrated that operational applications of glyphosate for forestry are unlikely to pose significant risk of harm because the concentrations found in forest water bodies after operational spraying are generally less than calculated or predicted concentrations (Thompson et al. 2004; Wojtaszek et al. 2004). The points of contention are the calculation or estimation of "realistic concentrations" for forest-use scenarios and the reliability of inferences from highly controlled laboratory or microcosm experiments when extrapolated to real-world aquatic environments. Three recent field studies under realistic exposure conditions show little or no harmful effects on amphibians (Edge et al. 2011, 2012*a*, 2012*b*). These field studies support suggestions by Thompson et al. (2006) and Thompson and Solomon (2010) that the weight of evidence from forestry-relevant studies with realistic test concentrations indicates that glyphosate will pose little risk of harmful effects on amphibians in the environment. However, others have disagreed (Relyea 2006, 2012).

2.1.2.3. Macroinvertebrates

By design, forest insecticides are intended to kill invertebrates and therefore aquatic macroinvertebrates could be at risk from insecticide applications. However, the forest insecticide Btk (distinct from its agricultural counterpart *Bacillus thuringiensis* var. *isrealensis* or Bti) is considered nontoxic to aquatic macroinvertebrates, and several forestry-relevant studies have demonstrated its environmental safety for aquatic macroinvertebrates (Table 1). Similarly, the molt-accelerating insecticide tebufenozide does not appear to pose a risk of harm to aquatic macroinvertebrates at or near expected concentrations (Kreutzweiser et al. 1994*c*; Song et al. 1997). All six of the forestry-relevant studies we found for the two main forest herbicides indicated that neither is likely to have significant adverse effects on aquatic macroinvertebrates at realistic concentrations (Table 1).

2.1.2.4. Zooplankton

Because of its unique insecticidal mode of action, Btk is not expected to be toxic to aquatic crustaceans, including zooplankton, and we could find no studies on the impacts of Btk on zooplankton. At least two studies have assessed the potential for adverse effects of the insecticide tebufenozide on zooplankton communities in a forestry context. One field experiment demonstrated significant adverse effects at realistic concentrations but under seminatural conditions (Kreutzweiser and Thomas 1995), while the other did not detect significant adverse effects under more natural conditions (Kreutzweiser et al. 1998*b*). The discrepancy points to some uncertainty around the risk estimation and suggests that efforts should be made to avoid contamination of water bodies by tebufenozide.

Few forestry-relevant or field studies have assessed the effects of common-use forest herbicides on zooplankton. A laboratory study suggested that triclopyr ester could be toxic to zooplankton at realistic concentrations (Chen et al. 2008). A similar laboratory study also determined that glyphosate could be harmful to zooplankton at expected concentrations (Chen et al. 2004), but other field-based studies indicated the risk was small under realistic conditions (Table 1). Wojtaszek (2004) measured the effects of these two herbicides on zooplankton communities and found that both caused reductions in some taxa and a shift in community structure at realistic concentrations, although the effects occurred at lower concentrations for triclopyr ester than for glyphosate.

2.1.2.5. Phytoplankton and periphyton

Common-use forest insecticides (and most insecticides in general) are not known to express phytotoxicity, so direct toxic effects on algal communities are not expected. Secondary effects on phytoplankton or periphyton can result from insecticide-induced reductions in algal grazers (e.g., Kreutzweiser and Thomas 1995) and therefore any forest insecticide that reduces (or increases) populations of grazers can affect phytoplankton communities. Herbicides by design are phytotoxic and can have adverse effects on phytoplankton and periphyton at realistic concentrations (Roshon et al. 1999). Forestry-relevant or field studies assessing the impacts of triclopyr ester or glyphosate on phytoplankton or periphyton are scarce, but a few field-based studies found no harmful effects on algal communities (Table 1). Rather, when the herbicides were toxic to zooplankton grazers there were often reciprocal short-term increases in phytoplankton or periphyton.

2.1.3. Forest management – prognosis

Because of the strong ecological linkages between forest water bodies and their surrounding watersheds, it is clear that large-scale forest harvesting can disrupt various land–water linkages and there can be measurable impacts on aquatic biodiversity as a result. However, the majority of studies in the boreal forest under recent or current forest management practices reported few adverse effects on aquatic organisms. Many studies were of relatively short duration (a few years), but they tended to show that when impacts did occur, there was usually evidence of a recovery trend within about five years. Improving forest management regulations and practices (Ice et al. 2010), forest stewardship certification schemes, and increasing recognition of water resource conservation through industry-supported programs, such as the Canadian Boreal Forest Agreement (Wells et al. 2010), should reduce the risk of forest harvesting impacts on aquatic biodiversity across the boreal zone.

However, there are some uncertainties that remain. We briefly discuss six issues for which uncertainties exist, but there are undoubtedly others. Some of these issues are also relevant to discussions about the impacts of other types of resource development, but we focus on forest management here. First, the impacts of forest harvesting at local or small-scale levels are well studied, such as effects on small streams, stream reaches, ponds, and individual lakes, but comparatively little is known about the potential for cumulative impacts and legacy effects on biodiversity across larger and multiple watersheds and landscapes. Small localized impacts may accumulate spatially and temporally over larger scales and potentially affect beta (landscape) diversity (Gustafson et al. 2007). This may become increasingly relevant as forest harvesting operations potentially expand into previously unmanaged landscapes, and overlap with other resource developments.

Secondly, most studies were focused on the types of aquatic environments with which aquatic ecologists are generally most familiar, such as distinct flowing stream channels, discrete ponds and lakes, and classified wetlands. But many aquatic systems across the boreal zone are more obscure, often owing to low topography and poor, unpredictable drainage patterns. These include cryptic (sometimes intermittent) headwaters and wetlands, "swampy," meandering, and silt-laden channels, and small, boggy ponds. These systems may also support ecologically important biodiversity but few studies have assessed harvesting impacts in those environments.

Thirdly, although each study defined reference or baseline conditions to which post-harvesting measurements were compared, ecologically relevant benchmarks or conservation targets were often not clear, especially when the benchmarks were influenced by climatic variability. When is a change construed as an impact, and when does the "impact" become unacceptable? To adequately assess impacts of forest management on aquatic biodiversity, it will be necessary to explicitly develop and define meaningful conservation targets for aquatic ecosystems in the boreal zone (Richardson and Thompson 2009).

Fourthly, the role of forest management (harvesting in particular) in accelerating or mitigating widespread calcium depletion from soil acidification in a changing climate and resultant impacts on aquatic organisms across North America has recently been identified as an uncertainty. Forest biomass harvesting and rapid uptake through revegetation following multiple harvest cycles have been included among the causes of declining lake–water calcium concentrations. Low calcium concentrations have been linked to near extirpations of important crustacean zooplankton species (*Daphnia* spp.), and up to 80% of 3700 lakes surveyed in Ontario on the Canadian Boreal Shield are approaching or are below the threshold at which these zooplankton species are at risk (Neary et al. 1990; Keller et al. 2001; Jeziorski et al. 2008). Although forest harvesting has been implicated in lake water calcium declines (Watmough et al. 2003, 2005), the extent to which forest management contributes to these declines in comparison to effects of soil acidification alone is unknown.

The fifth issue is the burgeoning use of forest biomass as a source of biofuels (Thiffault et al. 2010), which may pose risks to aquatic ecosystems. Forest harvesting regulations for the conservation of water resources are likely to apply to biomass harvesting, but their effectiveness is untested and unknown in boreal watersheds where biomass harvesting could include extraction of noncommercial trees, shrubs, and woody debris in operations and configurations that differ from commercial tree harvesting. Silvicultural and engineering approaches to increasing wood production from forest stands for biofuels (e.g., weed control, fertilization, drainage; Foster and Mayfield 2007) could lead to substantial and significant changes in water quality with associated effects on aquatic biodiversity. Forest biomass harvesting could cause further disturbances to boreal forest watersheds over and above commercial timber harvesting, especially with increasing pressure to venture into more northerly or previously inaccessible portions of the boreal forest for bioenergy feedstock.

A final uncertainty we address emerges from forestry jurisdictions across the boreal forest that are increasingly embracing ecosystem-based forest management and the use of natural disturbance emulation to conserve biodiversity (Long 2009). Natural disturbance emulation will have implications for conservation of aquatic biodiversity because water bodies are ubiquitous across, and are integral parts of, forest landscapes. Under this emerging paradigm, increased forest harvesting near water will be encouraged to emulate natural shoreline disturbances and increase shoreline habitat complexity (Naylor et al. 2012) and to potentially contribute to the long-term sustainability of aquatic ecosystems. An integration of our current understanding of land–water linkages with general disturbance ecology would suggest that periodic large-scale forest disturbances may be required for long-term sustainability of aquatic ecosystems in forest watersheds (Kreutzweiser et al. 2012), but this hypothesis is largely untested in the boreal forest and should be the focus of future research efforts (Sibley et al. 2012).

A large body of scientific evidence is available for assessing the impacts of forest pesticides on aquatic biodiversity and it strongly suggests that common-use forest pesticides pose little risk of harm to most aquatic organisms. A possible exception is the unresolved potential for harmful effects of the formulated herbicide glyphosate on amphibian populations, although there is considerable evidence that exposure to glyphosate under realistic, operational forestry conditions poses little risk to amphibians. Risks to aquatic biodiversity overall will be further reduced as pest managers increasingly implement improved management practices including the use of biological or other reduced-risk pesticides, the employment of no-spray buffer zones when appropriate, the use of advanced spray drift reduction systems, the use of improved spray guidance technologies, and the use of integrated pest management strategies in which the judicious application of pesticides is only one of several concurrent methods to control or manage losses from pest damage (Thompson et al. 2009; Kreutzweiser and Sibley 2012). As new forest pesticides are developed, it will be imperative that concomitant environmental assessment studies are conducted to ensure and demonstrate their environmental safety.

2.2. Pulp and paper effluents

Another potential hazard to aquatic biodiversity that is related to the forest industry, but at a different scale, is the discharge of effluents from forest product manufacturing, especially pulp and paper. In 2011, there were 17 active and 10 inactive pulp and paper mills across Canada's boreal zone (see Fig. 3 in Brandt et al. 2013). Whereas forest harvesting, roads, and pesticides potentially affect aquatic organisms across a broad spatial scale, the effects of pulp and paper effluents are restricted to mill discharge vicinities and some distances downstream. In response to scientific studies showing significant adverse effects of effluents in aquatic systems (e.g., special issue of the *Journal of the Fisheries Research Board of Canada* introduced by Kelso et al. 1977), strict regulations on effluent treatments were invoked across Canada in the early 1990s to limit the release of toxic byproducts (e.g., nutrients, dioxins, and furans) into receiving waters. The regulations also imposed an Environmental Effects Monitoring (EEM) program that required pulp mills to monitor the health of downstream aquatic ecosystems, with emphasis on the health of fish and macroinvertebrate communities. A large body of literature has reported and synthesized the results of the monitoring and research studies that resulted from this program. Our review focuses on recent trends under these newer regulations, but includes some pre-EEM findings to set the context for trends. Earlier reviews by Owens (1991), Munkittrick et al. (1998), Lowell et al. (2005), McMaster et al. (2006), and Kovacs et al. (2010) provide a more comprehensive look at this issue.

2.2.1. Fish

Anomalies in fish morphology, reproductive capacity, and population levels were the harbingers of the adverse effects of toxic constituents in pulp and paper mill effluents in the 1970s. Our assessment of 24 studies or reviews reporting the impacts of effluents on fish indicates that reductions in toxic discharge under the stricter regulations and the EEM have reduced the overall adverse effects of pulp and paper mill effluents on fish and fish populations (Table 2). Before the EEM program and discharge regulations were implemented, studies focused on fish survival, distribution, and community structure and showed that effluents often (but not always) caused reductions or shifts in these attributes. As refinements to effluent treatments and discharges were implemented and receiving water conditions improved, the risk of direct toxicity to fish was reduced and the focus shifted to measures of reproduction, morphology, organ condition, and sex hormones as population health indicators. Longer-term studies that spanned the periods before and after the EEM implementation generally showed improvements in these fish population indicators as well (Table 2). However, most also showed that several fish health indicators continued to be significantly impaired in comparison with upstream or reference conditions. The most recent studies show trends of improving fish population health in discharge areas, but some measurable impacts still exist (Bowron et al. 2009; Barrett et al. 2010; Kovacs et al. 2010). Although primary toxins (furans, dioxins) in pulp and paper effluents have been greatly reduced in recent years (Wiegand et al. 2006), continuing elevated nutrients and suspended solids can combine with low levels of toxins to produce measurable effects on fish. Elevated nutrients have resulted in increased fish growth, but this change has happened in conjunction with disrupted metabolic cycles and decreased gonad sizes that indicate persistent, sublethal effects of endocrine-disrupting toxins on reproduction (Lowell et al. 2005).

2.2.2. Amphibians

We found no published studies that directly measured effects of pulp and paper effluents on amphibians. Some of the minor toxins that have traditionally been released from mills (e.g., phenols) can be toxic to amphibians at realistic concentrations (Buikema et al. 1979) but at least two assessments have determined that the likelihood that phenol concentrations would be significant enough to harm amphibians in discharge areas is low (Fox 2001; Breton et al. 2003). Nevertheless, it seems reasonable to assume that fish and aquatic larval stages of amphibians are likely to be similar in terms of their sensitivity to the toxic compounds in pulp effluents and therefore studies to measure potential effects on amphibian development, growth, and reproduction are warranted.

2.2.3. Macroinvertebrates

From the earliest studies, it was apparent that pulp mill effluents could be detrimental to aquatic macroinvertebrate communities in discharge areas and downstream vicinities (Vander Wal 1977). Because of the sensitivity, ubiquity, and ecological significance of such communities, a requirement for aquatic macroinvertebrate monitoring was formally included in the EEM, and since then this monitoring has been a useful tool for assessing environmental impacts of mill discharges. Studies and reviews conducted since the implementation of this EEM requirement demonstrate that improved water quality conditions at discharge areas following enhanced effluent treatments have reduced the effects on macroinvertebrate communities in terms of direct toxic effects and population declines (Table 2), similar to the responses seen in fish communities. Of the seven studies since 2000 that we reviewed, all still reported significant or measurable effects on macroinvertebrate communities but the most common impacts were related to eutrophication (nutrient enrichment) in discharge areas (Chambers et al. 2006). Nutrient and organic content enrichment from mill discharges tends to cause increased primary production that subsequently results in secondary increases in macroinvertebrate abundance and (or) shifts in community structure. These changes can constitute harmful effects on macroinvertebrate communities as they can disrupt food webs and normal ecosystem functioning (Sibley et al. 2001; Culp et al. 2000*a*).

2.2.4. Zooplankton

Risks to zooplankton are generally assessed by standardized bioassays with pulp and paper effluents and common micro crustacean test species, such as *Daphnia* spp., and are included as part of the assessment of toxicity to invertebrates. Therefore, none of the studies we reviewed reported measurements of impacts on zooplankton communities in situ. The toxic compounds in many mill effluents can impair *Daphnia* survival, growth, or reproduction (Table 2), but the risk of adverse effects in receiving waters is declining as the major toxins in effluents decline. Recent studies show that some effluents can still cause harmful effects on zooplankton test species, but overall, improved effluent treatments continue to reduce the risk of harm to zooplankton and other invertebrates (Kovacs et al. 2010).

2.2.5. Phytoplankton and periphyton

Pulp mill discharges can produce eutrophication effects from increased nutrients and organic compounds that clearly affect phytoplankton and periphyton communities. Eight of the 10 studies or reviews we examined reported increased biomass and (or) shifts in community structure in discharge areas or experimental settings (Table 2). Eutrophication effects from pulp mill effluents cannot always be separated from those arising from municipal effluents in the same watersheds (Scrimgeour and Chambers 2000) but nonetheless pose a risk of harm to natural primary production in boreal watersheds. Schindler and Lee (2010) point out that eutrophication in boreal lakes is an increasing problem, but they suggest that the contribution of pulp mill effluents to that problem is minimal in comparison to that of municipal, agricultural, and shoreline development. As indicated earlier, these types of changes in biomass and community structure at the base of food webs, particularly blooms of nuisance algae, can constitute harmful effects on overall ecosystem functioning (Chambers et al. 2006). Although we did not find published studies that assessed impacts of effluents on macrophytes in boreal waters, other studies that included marine or estuarine plant communities showed similar eutrophication effects (increased biomass, altered plant communities) where nutrient concentrations exceeded natural levels (Zimmerman and Livingston 1976).

2.2.6. Pulp and paper effluents – prognosis

Improvements in pulp mill effluent treatments in compliance with toxin-control regulations over the past couple of decades have drastically reduced toxin concentrations in receiving waters and improved water quality overall near discharge areas. Water quality improvement is an issue that the pulp and paper industry seems to have taken seriously. Industry and the Canadian government have recently combined efforts to improve the environmental performance of pulp and paper production under Natural

Table 2. Impacts of pulp and paper effluents on aquatic biodiversity.

Indicator	Disturbance type or timing of study	Location	Length of study	Assessment end point	Impact*	Reference
Fish	Before EEM†	Ontario	Short term	Fish health (organic contaminant residues in tissues)	-	Kaiser 1977
	Before EEM†	Ontario	Short term	Density; distribution; movement	-; –; - (shift in species dominance)	Kelso et al. 1977
	Before EEM†	Ontario	1 year	Distribution; survival	0; -	Leslie and Kelso 1977
	Primary or secondary effluent	Ontario	Varies	Fathead minnow survival; larval growth	– or 0; 0 (stream and treatment dependent)	Robinson et al. 1994
	Primary or secondary effluent, dioxins and furans	Ontario	2 months	Health	0 (no correlation between ↓ sex steroids and ↑ toxins)	Servos et al. 1994
	Primary or secondary effluent	Saskatchewan	2 years	Fish populations (walleye and white sucker): growth; health; fecundity	0; -; 0	Swanson et al. 1996
	Primary or secondary effluent	Ontario	Short term	Rainbow trout survival; growth	-; –	Whittle and Flood 1977
	Before and after EEM†	Canada (not all boreal)	12 years	Body condition; gonad size	++; – (before) +; - (after)	Barrett et al. 2010
	Before and after EEM†	Ontario	20 years	Health	– (before) - (after)	Bowron et al. 2009
	Before and after EEM†	Canada (not all boreal)	Report, 10 years	Reproduction	– (before) - (after)	McMaster et al. 2006
	Before and after EEM†	Global (not all boreal)	Review	Reproduction	– (before) - (after)	Munkittrick et al. 1998
	After EEM†	Alberta	Report	Population health; sex hormones	-; –	Cash et al. 2000
	After EEM†	Canada (not all boreal)	Report, 10 years	Gonad size; body condition	-; +	Environment Canada 2003
	After EEM†	Global (not boreal)	Review	Reproduction	-	Hewitt et al. 2008
	After EEM†	Canada	12 years	Body size; reproduction	+; -	Lowell et al. 2005
	After EEM†	Canada (not all boreal)	Varies	Egg survival; larval survival; egg production; sex ratio	0; 0; -; -	Kovacs and Megraw 1996
	After EEM†	Canada (not all boreal)	Review	Rainbow trout survival	- or 0 (dependent on mill)	Kovacs et al. 2004
	After EEM†	Canada (not all boreal)	Short term	Fathead minnow reproduction	-	Kovacs et al. 2005
	After EEM†	Global (not all boreal)	Review, 5 years	Reproduction	-	Munkittrick et al. 1997
	Secondary effluent chlorine substitution	Ontario	2 months	Health (hormones, organ size, physiology)	-	Munkittrick et al. 1994
	Secondary effluent	Canada (not all boreal)	Review	Rainbow trout survival	- or 0 (dependent on mill)	O'Connor and Voss 1998
	Secondary effluent	Quebec	One time sample and laboratory	Survival; organ size; growth; egg production	0; +; - or +; +	Parrott et al. 2010
	Secondary effluent	Canada	Review	Reproductive health; community composition	- or 0; - or 0 (dependent on mill)	Kovacs et al. 2010
Macroinvertebrates	Before EEM†	Quebec	3 months	Abundance; richness	-; –	Hilton 1980
	Before EEM†	Global (not all boreal)	Review	Diversity	- (primary effluent) 0 (secondary effluent)	McLeay 1987
	Before EEM†	Ontario	Short term	Diversity; distribution	–; –	Vander Wal 1977
	After EEM†	Alberta	1 year	Abundance; richness	+; 0	Culp et al. 2000*a*
	After EEM†	British Columbia (not boreal)	Review, 6 weeks	Richness; biomass	0; +	Culp et al. 2000*b*
	After EEM†	Canada (not all boreal)	Report, 10 years	Abundance; richness	+; 0	Environment Canada 2003
	After EEM†	Canada	12 years	Abundance; richness	+; 0 or - or +	Lowell et al. 2005
	After EEM†	British Columbia and Alberta	20 years and a few weeks	Abundance	+, 0 (nutrient–toxicity tradeoff)	Lowell et al. 2000

Table 2 (*concluded*).

Indicator	Disturbance type or timing of study	Location	Length of study	Assessment end point	Impact*	Reference
	Effluent, decreased oxygen	Alberta	2 weeks	Mayfly survival	–	Lowell and Culp 1999
	Effluent	British Columbia	2 weeks	Mayfly size	+	Lowell et al. 1995
	After EEM†	Canada	Report, 10 years	Abundance; richness	+; 0	McMaster et al. 2006
	After EEM†	Alberta	28 days	Community composition; growth	0; + or 0 (species dependent)	Podemski and Culp 1996
	After EEM†	Ontario	1 year	Density; community composition; diversity	0; -; -	Sibley et al. 2001
Zooplankton	Before EEM†	Ontario	Short term	*Daphnia* sp. health inferred from filtering rate performance	-	Cooley 1977
	Bleached effluents	Global (not all boreal)	Varies	*Daphnia* reproduction	– or - (dependent on effluent)	Kovacs and Megraw 1996
	Primary or secondary effluent	Ontario	Varies	*Daphnia* survival; reproduction	0; - or + (primary) 0 or + (secondary)	Robinson et al. 1994
	After EEM†	Canada (not all boreal)	Review	*Daphnia* survival	- or 0 (dependent on mill)	Kovacs et al. 2004
	After EEM†	Canada (not all boreal)	Review	*Daphnia* survival	- or 0 (dependent on mill)	O'Connor and Voss 1998
Phytoplankton and periphyton	Before EEM†	Ontario	Short term	Periphyton; phytoplankton productivity	–; –	Moore and Love 1977
	Primary effluent	Global (not all boreal)	Review	Phytoplankton diversity; biomass	-; +	McLeay 1987
	After EEM,† effluent	Alberta	10 years	Periphyton production	+	Chambers et al. 2006
	After EEM,† Effluent	Alberta	1 year	Periphyton biomass	+	Culp et al. 2000*a*
	After EEM,† effluent	British Columbia (not boreal)	Review, 6 weeks	Periphyton densities; biomass	-; +	Culp et al. 2000*b*
	Bleached effluents	Global (not all boreal)	Varies	Algal growth	-	Kovacs and Megraw 1996
	Effluents	British Columbia and Alberta	20 years and a few weeks	Algal growth; biomass	+; +	Lowell et al. 2000
	Effluents	Alberta	28 days	Periphyton community	+	Podemski and Culp 1996
	Mill and sewage effluents	Alberta	13 years	Periphyton biomass	+	Chambers et al. 2000
	Mill and sewage effluent	Alberta	1 year	Periphyton biomass	++	Scrimgeour and Chambers 2000

*The impact rating scale is as follows: –, large decrease; -, moderate decrease; 0, little or no measurable effect (where little is either a small or a brief change); +, moderate increase; and ++, large increase. End points are listed together, separated by a semicolon, for studies that examined multiple assessment end points.

†EEM, Environmental Effects Monitoring. The EEM program was implemented in 1992, with a deference period that extended to 1996. The objective was to monitor mitigation efforts aimed at reducing the detrimental effects of effluents on aquatic systems. Before the program was introduced, most mills only conducted primary treatment and therefore effluents often retained high concentrations of toxins. Following implementation of EEM, secondary effluent treatment became mandatory and all mills were required to monitor the health of the aquatic environment receiving effluents.

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Resources Canada's Pulp and Paper Green Transformation Program (Natural Resources Canada 2010). As treatment technologies evolve, water quality in receiving areas is expected to improve further. However, not all adverse effects from pulp mill effluents have been eliminated. Some recent studies have continued to show measurable harmful effects, mostly from eutrophication or sedimentation, and mostly dependent on the mill's production technology, treatment technology, and dilution potential at discharge sites. Increased focus on technologies to further reduce nutrients and suspended solids is warranted. Potentially confounding influences, such as effluents from other sources (notably municipal and agricultural sources), and reduced receiving water volumes from climate change - induced droughts and water extractions that affect discharge dilution potential, will increasingly challenge effluent management programs. Watershed-level monitoring of these conditions at mill sites (see Fig. 3 in Brandt et al. 2013) will assist in improving effluent management efforts. If the trend of declining pulp and paper demand seen in the past few years continues and production across the boreal forest continues to decrease as a result, risks to aquatic biodiversity from harmful effects of mill effluents will decline accordingly, but may still be significant at local or watershed scales.

2.3. Hydroelectric impoundments

Dams, impoundments, and water level control on regulated rivers have profound effects on aquatic ecosystems and their biodiversity. Although much of what is known about the impacts of river regulation on aquatic biota is from studies outside of the Canadian boreal zone, a number of recent studies conducted in or near boreal watersheds have contributed to that knowledge base. Direct, and often obvious, impacts of dams and impoundments include flooding of riparian, wetland, and upland habitats, conversion of lotic (flowing) environments to lentic (standing water) systems, creation of lakes or reservoirs, submersion of large organic matter pools, shoreline erosion and increased sedimentation, disruption of natural hydrographs, fluctuating water levels above and below dams, downstream scouring, increased turbidity, redirected river-channel morphology, changes in biogeochemical cycling, and alterations to natural water temperature patterns. Most regulated rivers in the boreal zone are dammed for generation of hydroelectric power (see Fig. 9 in Brandt et al. 2013), and this imposes further disturbances on aquatic systems from the creation of clear-cut transmission corridors through forest wetlands and headwaters that can interfere with natural hydrological patterns. Any or all of these have implications for the survival, production, and movement of aquatic organisms (Rosenberg et al. 1997; McAllister et al. 2000; Urquizo et al. 2000; Cott et al. 2008*a*; Scruton et al. 2008; Poff and Zimmerman 2010; Smokorowski et al. 2011; Bajzak and Roberts 2011). The flooding of organic-rich boreal forest floors increases methylation and the release of natural mercury and greenhouse gases like methane, CO_2, and NO_x (Webster et al. Manuscript in preparation). Increased methylmercury concentrations in sediments and water can directly affect fish health (Latif et al. 2001; Larose et al. 2008), but such increases are primarily a human health issue because methylmercury accumulates in food webs and poses serious risks to human health when people consume predatory fish (Rosenberg et al. 1997; Diez 2009). See the review by Yang et al. (2008) for further information on methylmercury contamination and its implications.

2.3.1. Fish

The damming of boreal rivers for hydroelectric production clearly has effects on native fish populations. Seventeen of the 18 studies or reviews that we examined, which included over 100 studies in boreal and nonboreal regions, reported adverse effects on at least some of the measurement end points for impacts on fish (Table 3). Adverse effects included reductions in community structure, diversity, abundance, spawning habitat, and reproduction of various fish species. These were often related to the loss of habitat or spawning beds from water drawdowns and reduced flows, habitat degradation from sedimentation or downstream scouring, blocked movement of migratory species, increased turbidity, changes in temperature, and physiological stress that the fish experience in dealing with fluctuating hydraulic conditions. The extent and magnitude of most impacts were almost always correlated with the magnitude, duration, and timing of flow alterations. Other effects that were occasionally reported included increased habitat for lentic and warm-water species, increased food resources from terrestrial inputs, and increased production of some species. River fragmentation by construction of hydroelectric dams has been shown to adversely affect fish populations, to the extent of extirpation of species over broad areas, in nonboreal studies (Rosenberg et al. 1997; Nilsson et al. 2005), but we were unable to find studies from the North American boreal zone that specifically assessed this threat. A couple of recent studies tended to show fewer adverse effects from hydroelectric dams than most previous studies (Marty et al. 2009; Smokorowski et al. 2011), suggesting that newer regulations on water fluctuations (ramping rates) may be reducing some of the risk of harmful effects to fish. However, it appears that new ramping rate regulations to emulate natural flow patterns are often voluntary or applied on a case-by-case basis to address local concerns, rather than under any comprehensive or national regulation framework (Marty et al. 2009).

2.3.2. Macroinvertebrates

Macroinvertebrate responses to the effects of river damming have been more variable. There is no question that as benthic and littoral habitats change above and below impoundments, invertebrate communities will also change. Assessment of impacts on macroinvertebrates varied depending on the end points being measured and the locations of sampling. Invertebrate communities in thalweg (midstream) positions responded differently than those along edges of channels and impoundments (Smokorowski et al. 2011). Studies reported reduced diversity and biomass, no measurable impacts, and increased diversity, biomass, and abundance (Table 3). The combined results of the studies indicated that local macroinvertebrate communities (i.e., those near dams) are likely to change in response to the introduction of dams and impoundments; this change is likely to involve a shift from flow-dependent and sediment-intolerant species to more lentic and sediment-tolerant assemblages. This will alter the natural riverine invertebrate communities for that particular area, and the extent to which those alterations occur will reflect the extent to which the changed habitat conditions (flow, sediments, temperatures, water quality) infringe on the natural river channels. Some of those areas can be large. The Smallwood Reservoir on the Churchill River of Newfoundland and Labrador alone is over 60 km long and covers an area of about 6500 km². Although natural invertebrate communities at that scale are undoubtedly altered, there was no indication from the combined results of the studies that any particular invertebrate species or assemblages were threatened at a regional scale by hydroelectric dams across the boreal zone. Changes in reservoir and downstream invertebrate communities are related to the magnitude of fluctuations in water level and flow, and new regulations that impose flow constraints on hydroelectric dam operations to more closely emulate natural flow regimes can reduce adverse effects on downstream invertebrates (Patterson and Smokorowski 2011; White et al. 2011).

2.3.3. Zooplankton, phytoplankton, and periphyton

The creation of hydroelectric impoundments drastically increases the lentic (standing water) environments in watersheds and introduces vast amounts of terrestrially derived nutrients and

Table 3. Impacts of hydroelectric impoundments on aquatic biodiversity.

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
Fish	Impoundments	Canada	Review	Community structure; migration	-; -	Browne 2007
	Impoundments	Canada	Review	Abundance	+ or –(species dependent)	Legault et al. 2004
	Impoundments	Global (not all boreal)	Review	Migration; diversity	-; -	McAllister et al. 2000
	Impoundments	Global (not all boreal)	Review	Abundance; diversity	-; -	Poff and Zimmerman 2010
	Impoundments	Global (not all boreal)	Summary	Production; migration	+; - (short-term increase)	Schindler et al. 1990
	Impoundments	Canada	Review	Spawning habitat	–	Schindler 1998*a*
	Altered flow regime	Canada	Review	Quality of spawning habitat	-	Browne 2007
	Regulated flow	Ontario	5 months	Hyperactivity	+ (↑ with extreme flows)	Murchie and Smokorowski 2004
	Regulated flow	Ontario	3 years	Biomass; diversity; condition; food web	0; -; +; 0	Smokorowski et al. 2011
	Regulated flow	Ontario	4 years	Food web connectivity; web length	0; -	Marty et al. 2009
	Hydro-peaking	Newfoundland	Short term	Movement	+ (↑ with extreme flows)	Scruton et al. 2003
	Hydro-peaking	Newfoundland	Short term	Juvenile salmon behaviour	-	Scruton et al. 2005
	Water diversion	National	Review	Population abundance	–	Allan et al. 2000
	Water drawdown	Quebec	2 years	Reproduction, egg survival	– (shallow water) - (deep water)	Benoit and Legault 2002
	Water drawdown	National	Review	Growth; abundance; survival; recruitment	-; -; -; -	Cott et al. 2008*a*
	Water drawdown	Northwest Territories	2 years	Habitat; abundance	0 or -; 0 (% drawdown dependent)	Cott et al. 2008*b*
	Water drawdown	Canada	Review	Lake trout population; reproduction	-; 0 or – (% drawdown dependent)	Legault et al. 2004
	Water withdrawal	Canada	Review	Migration; diversity	-; -	Cunjak 1996
Macroinvertebrates	Impoundments	Global (not all boreal)	Review	Mollusc diversity	-	McAllister et al. 2000
	Impoundments	Global (not all boreal)	Review	Abundance; diversity	- or +; - or + (species dependent)	Poff and Zimmerman 2010
	Impoundments	Boreal Shield	Comparative study	Richness; functional composition	- or 0; - or 0 (fluctuation dependent)	White et al. 2011
	Water drawdown	National	Review	Biomass	+ or - (species dependent)	Cott et al. 2008*a*
	Water drawdown	Experimental, not boreal	3 years	Richness; density; biomass	0; -; –	Walters and Post 2011
	Altered flow regime	Ontario	9 weeks	Abundance; diversity; feeding guild proportions	0; -; -	Patterson and Smokorowski 2011
	Altered flow regime	Ontario	3 years	Abundance; diversity	0; 0 or - (species, location dependent)	Smokorowski et al. 2011
	Flooding	Ontario	5 years	Chironomidae emergence (two habitat types)	–; ++	Rosenberg et al. 2001
Zooplankton	Impoundments	Ontario	3 years	Biomass	++	Paterson et al. 1997
	Impoundments	Global (not all boreal)	Summary	Biomass	-	Schindler et al. 1990
	Impoundments	Canada	Review	Abundance; biomass; community structure	0; 0; -	Legault et al. 2004
	Water drawdown	Ontario	Review	Community composition	0	Cott et al. 2008*a*

Table 3 (*concluded*).

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
Phytoplankton and periphyton	Impoundments	Canada	Review	Abundance	0 (short-term increase)	Legault et al. 2004
	Impoundments	Ontario	3 years	Phytoplankton production; phytoplankton biomass	-; -	Paterson et al. 1997
	Impoundments	Ontario	200 years	Diatom assemblage	0	Ruhland et al. 2010
	Water drawdown	Ontario	Review	Algal biomass; phytoplankton community	-; 0	Cott et al. 2008*a*
	Water drawdown	Ontario	3 years	Phytoplankton biomass; assemblages; productivity; diversity	0; 0; 0; -	Turner et al. 2005
Macrophytes	Water drawdown	Ontario	Review	Abundance	+	Cott et al. 2008*a*
	Flooded shallow wetland	Ontario	9 years	Vegetation community structure (stability)	- (shift to bog species)	Asada et al. 2005
	Water drawdown	Ontario	3 years	Biomass; cover; richness	-; -; -	Turner et al. 2005

*The impact rating scale is as follows: --, large decrease; -, moderate decrease; 0, little or no measurable effect (where little is either a small or a brief change); +, moderate increase; and ++, large increase. End points are listed together, separated by a semicolon, for studies that examined multiple assessment end points.

organic matter to those environments. These changes influence the relative abundance and biomass of planktonic and algal species. Relatively few studies have assessed the impacts of hydroelectric impoundments on plankton and algae in boreal water bodies, but among those that did, responses were variable, specific to particular species or functional groups, and dependent on other factors, such as turbidity and light level (Legault et al. 2004). The studies and reviews we examined tended to show minimal changes in these communities overall, although measurable increases and decreases among some metrics for certain groups were reported in six of the nine studies we found (Table 3).

2.3.4. Macrophytes

Very few studies have been conducted on Canadian boreal watersheds to determine the effects of hydroelectric impoundments on aquatic macrophytes. However, a larger literature on the effects of water level fluctuations on aquatic plants in general demonstrates that macrophyte occurrence and growth will be proportional to water level fluctuations, water clarity, reservoir configuration or shape, littoral (near shore) conditions and slope, and exposure to wave action (Clayton and Champion 2006). Flooded, shallow, organic-rich areas are likely to be prolific environments for many aquatic macrophytes, whereas unstable, fluctuating, and scour-prone shorelines will be detrimental to the establishment and growth of aquatic plants. Accordingly, the three studies on boreal hydroelectric projects we found have reported both increases and decreases in macrophytes as a result of hydroelectric impoundments (Table 3). Turner et al. (2005) suggest that impacts on macrophytes will be greater in littoral zones than in pelagic (deeper) areas of reservoirs.

2.3.5. Hydroelectric impoundments – prognosis

Although there is a relative paucity of studies in Canadian boreal watersheds, those that have been reported combined with the broader literature on hydroelectric impacts unequivocally demonstrate that dams and impoundments have profound effects on aquatic ecosystems and their biodiversity. Aquatic habitat conditions are drastically altered by hydroelectric dams and these have variable, but emphatic, effects on aquatic organisms. If the size and frequency of hydroelectric installations increase across boreal watersheds, the effects on local, possibly even watershed-level, aquatic biodiversity will increase accordingly. Not all effects are harmful to aquatic biodiversity. Some are beneficial to certain organisms and assemblages, but all effects invoke large changes in biodiversity structure and potentially in some functional attributes of biotic communities. If the main goal of biodiversity conservation targets is that hydroelectric developments should induce no change beyond those that would be seen under natural conditions, then most or all installations will fall short of this goal. But if the goal is to sustain healthy aquatic ecosystems with representative natural biotic assemblages, then some mitigation principles can be used to guide the development and operation of hydroelectric installations.

To minimize the adverse effects of hydroelectric dams and impoundments on aquatic biodiversity, land use planners and managers should consider the following guiding principles. The complexity (variation of habitat types) of the overall aquatic ecosystem within watersheds should be retained, with installations interspersed to maintain some areas with natural lotic conditions and habitats. Dams should include passage facilities for fish and other aquatic organisms. Known biodiversity "hotspots" or critical habitats for endangered, threatened, or rare species should be avoided. Water level fluctuations should be minimized and regulated to mimic as closely as possible the natural flow regimes for a given watershed. The flooding of wetlands should be avoided whenever possible. The use of new technologies to produce power from smaller reservoirs or run-of-the-river operations should be encouraged, although Schindler and Lee (2010) point out some problems associated with these technologies. Dams that are no longer functioning should be carefully decommissioned. Finally, bioassessment and monitoring programs at each installation to measure the success of mitigation efforts and to facilitate an adaptive management approach to further development would be useful.

A related topic worth noting is thermal power generation across the boreal zone and its potential impacts on aquatic ecosystems. We found only two older publications that assessed or reviewed the impacts of thermal power generation on water resources in Canada (Gallup and Hickman 1975; Dickson 1976), but the recent report by the National Round Table on the Environment and the Economy (NRTEE 2010) points out that water extraction and use by that industry could be a continuing stressor on water quality in receiving areas. Thermoelectric power generation is the second largest water user among natural resource sectors in Canada (agriculture is by far the largest). There are about 55 thermal power generation stations across the boreal zone, mostly in Alberta and the Northwest Territories, but the number is expected to decline in favour of hydroelectric and other power generation sources (NRTEE 2010). The primary concerns associated with thermal power generation are elevated temperatures of discharge water and some potential pollution from corrosion-control products used in cooling water. Oil- and coal-fired plants generally use and discharge more water than plants fired by natural gas, and gas-fired plants are becoming more prevalent. The industry is responding to concerns about water quality and use by using improved technologies to reduce its reliance on water sources, but it is not clear to what extent these actions are effective (NRTEE 2010).

2.4. Mining

The extraction and processing of minerals and metals is scattered throughout Canada's boreal zone, especially in midlatitude and southern areas (see Fig. 6 in Brandt et al. 2013). As of 2009, there were 99 active mines, six smelters, and nine coal mines in the boreal zone. Water extraction and use, mine and site facility construction, and their associated road networks can alter the flows and condition of groundwater and surface water, but the primary risks to aquatic biodiversity from mining activities arise from turbidity and potentially toxic compounds that leach or are discharged from tailings and effluents (Urquizo et al. 2000). Potential discharge of toxic materials from active mines is an ongoing risk to aquatic ecosystems, but there are a growing number of abandoned mines under varying stages of decommissioning or remediation that may also pose a risk of long-term leakage of toxic materials (CESD 2002; CDL 2005; Cowan et al. 2010). A vast literature from nonboreal studies and laboratory toxicity tests demonstrates that elevated levels of metals, minerals, and solids in receiving waters and sediments pose a significant risk of harm to aquatic organisms at environmentally realistic concentrations (e.g., Gerhardt 1993; Kong et al. 1995; Bren 2001; Fisher and Hook 2002; Lydersen et al. 2002; Rainbow 2002; Norwood et al. 2003; Peijnenburg and Jager 2003; Witeska and Jezierska 2003; Pane et al. 2004).

Additionally, the smelting of metals has historically been a contributor to acid rain when sulphate and nitrate-based emissions combine with precipitation, and the resulting acidic deposition on watersheds can reduce pH in receiving waters to toxic levels. Acidic deposition is a transboundary pollution problem (Singh and Agrawal 2008), but in the Canadian boreal zone, areas of highest acidic deposition and susceptibility to acidification (lowest buffering capacity) tend to occur in the eastern Boreal Shield (Environment Canada 2005). However, parts of northeastern Alberta and northern Saskatchewan also have relatively low buffering capacity, and with increasing industrial development in that region, water bodies of the western boreal zone are at increasing risk (Aherne and Shaw 2010). Acidification of water bodies from airborne pollutants has severe and long-lasting adverse effects on aquatic biodiversity (Schindler 1988). Our review of impacts of mining effluents on aquatic biodiversity does not include a review of acid rain impacts, but a vast literature exists on the topic. Publications on acid rain impacts and recovery in the Canadian boreal zone include Carbone et al. (1998), Keller et al. (1992), Watt et al. (2000), Doka et al. (2003), Jeffries et al. (2003), Snucins and Gunn (2003), Clair et al. (2007), Keller et al. (2007), Gray and Arnott (2009), Keller (2009), Scott et al. (2010), and Valois et al. (2010).

Despite the prevalence of mining throughout the entire boreal zone, surprisingly few published studies were found that assessed the impacts of mining effluents or tailings on aquatic biota in Canadian boreal watersheds (Table 4). A notable exception is the concentration of published studies from the nickel-mining region near Sudbury, Ontario, where the recovery of aquatic ecosystems from mining impacts has been investigated for over 30 years (Gunn 1995; Gunn et al. 1995). However, in those studies, metal toxicity in lakes is closely coupled with smelter-induced acidification (LaZerte 1986) and the specific impacts of elevated metals are confounded and compounded by acid effects (Keller and Pitblado 1986; Keller et al. 2004).

In the early 1990s, the Government of Canada commissioned a multistakeholder working group to assess and report on the effects of metal mining on aquatic ecosystems (AQUAMIN 1996). That assessment found frequent instances of elevated metal concentrations in downstream reaches, lakes, and sediments and evidence of altered aquatic invertebrate communities at about half of the 95 study sites. The working group also reported that the assessment of effects on fish populations was meager, but that several studies reported significant reductions in fish abundance in the vicinities of mining operations. Documented instances of metal uptake and adverse effects on zooplankton and phytoplankton were also reported. The working group concluded that a number of impacts on aquatic organisms could be attributed to mining operations and effluent releases, that older mines and sites generally had more pronounced effects, and that a number of sites showed signs of improvement and recovery (AQUAMIN 1996). Industry requirements for monitoring and assessment of aquatic effects continue under Environment Canada's EEM program (Scroggins et al. 2002; Walker et al. 2003), and a recent assessment and meta-analysis of the program's performance concluded that significant and often inhibitory effects on fish and aquatic invertebrates are still reported under the EEM from many mining sites (Environment Canada 2012).

2.4.1. Fish

Thirteen of the 15 published studies and reviews relevant to the Canadian boreal zone that we examined reported significant adverse effects on various fish measurement end points (Table 4). It is difficult to make a general assessment of mining effluent effects on fish in the boreal zone, owing to the wide variation in the responses and response measurements, in the study conditions, in the types of mining and processing, and in the effluent treatments, constituents, and concentrations (e.g., Pyle et al. 2008). When mining effluents coincide with other effluent sources, it is not always clear the extent to which mining effluents are contributing to observed changes or impacts (Weber et al. 2008). Nevertheless, fish populations in receiving waters of mining effluents often showed significant adverse effects on survival or growth and on various sublethal indicators of population or individual health (Rasmussen et al. 2008). Studies that spanned several decades showed trends of recovery from highly decimated fish populations to stressed but recovering fish communities in metal-contaminated lakes (which were also acidified) (Lippert et al. 2007). Those long-term studies also demonstrated that the recovery of

Table 4. Impacts of minerals and metal mining on aquatic biodiversity.

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
Fish	Mining effluents	Saskatchewan	1 year	Juvenile survival; energy reserves	0; +	Bennett and Janz 2007
	Selenium in effluents	British Columbia	One collection	Egg survival; alevin survival; larval survival; growth; morphology	0; 0; 0; -; –	McDonald et al. 2010
	Selenium in effluents	Saskatchewan	2 years	Selenium bioaccumulation extrapolated to reproduction	-	Muscatello et al. 2008
	Selenium from effluents	Saskatchewan	One collection	Selenium in tissues; juvenile condition	+; 0	Muscatello and Janz 2009
	Metals in lake	Labrador	2 weeks	Growth hormone	–	Fahraeus-Van Ree and Payne 2005
	Metals in lakes	Ontario, Quebec	2 years	Longevity, condition	-; -	Pyle et al. 2008
	Iron-ore effluents	Labrador	3 years	Health (skin bleaching)	–	Payne et al. 2001
	Gold mine	Ontario	Review	Habitat; walleye spawning; growth	-; -; -	Browne 2007
	Gold mine tailings spill	Ontario	2 years	Survival of walleye eggs	–	Leis and Fox 1994
	Nickel mining	Ontario	Review	Fish reproduction	-	Browne 2007
	Mining effluents and municipal waste water	Ontario	1 year	Juvenile growth; energy storage	+ or -; + or 0 (species dependent)	Driedger et al. 2009
	Mining effluents and municipal waste water	Ontario	2 months	Egg production; embryo health	+; –	Rickwood et al. 2008
	Mining effluents	Ontario	3 weeks	Egg production	-	Rozon-Ramilo et al. 2011
	Metals in lakes (acidified)	Ontario	1 year	Organ health; population; growth	-; –; –	Rasmussen et al. 2008
	Metals in lakes (acidified)	Ontario	30 years	Abundance; diversity	-; – (some recovery)	Lippert et al. 2007
Macroinvertebrates	Acid mine drainage diversion	Ontario	9 years	Taxa richness; diversity	–; –	Gunn et al. 2010
	Metal mining effluents	Northwest Territories	1 year	Abundance; richness; diversity;	0 or +; 0 or +; 0 or +	Spencer et al. 2008
	Metal mining effluents	Ontario	2 months	Midge survival; emergence; hatching success; growth; reproduction	-; -; -; 0; 0	Hruska and Dube 2004
	Metals in lakes (acidified)	Ontario	Survey	Littoral invertebrate abundance; diversity	–; – (some recovery)	Wesolek et al. 2010
Zooplankton	Metals in lakes (possibly acidified)	Saskatchewan	10 years	Community composition; diversity; abundance	-; -; –	Melville 1995
	Metals in lakes (acidified)	Ontario	30 years	Copepod population recovery; cladoceran population recovery	+; –	Yan et al. 2004
	Metals in lakes (acidified)	Ontario	30 years	Abundance; diversity	-; – (some recovery)	Valois et al. 2010
Phytoplankton	Metals in lakes (acidified)	Ontario	5 years	Community structure; biomass	–; -	Yan 1979
	Metals in lakes (acidified)	Ontario	30 years	Diversity; metal tolerance by species	–; -	Woodfine et al. 2002
	Mine effluent into lakes	Northwest Territories	2 years	Abundance	-	Moore et al. 1979
	Flooded mine site	Saskatchewan	7 years	Community composition	–	Kalin et al. 2001
	Acid mine drainage	Ontario	17 years	Diversity; richness	–; - or + (some recovery)	Kalin et al. 2006

*The impact rating scale is as follows: –, large decrease; -, moderate decrease; 0, little or no measurable effect (where little is either a small or a brief change); +, moderate increase; and ++, large increase. End points are listed together, separated by a semicolon, for studies that examined multiple assessment end points.

fish communities from metal contamination can take decades when bioavailable metal concentrations persist at or near toxic thresholds, especially when coupled with acidification effects (LaZerte 1986).

2.4.2. Macroinvertebrates

Aquatic invertebrate communities are recognized by the working group on aquatic effects of mining (AQUAMIN 1996), the regulatory EEM program in Canada (Scroggins et al. 2002; Walker et al. 2003), and the broader scientific community (Bren 2001) as effective indicators of effluent effects on the aquatic environment, but we found only four recently published studies that have assessed and reported those effects in Canadian boreal waters (Table 4). Three of those reported impacts that clearly show negative and long-lasting effects on invertebrate communities, especially reduced diversity, when receiving waters and sediments contain bioavailable metals. Recent studies in metal-contaminated lakes or streams continue to show lower invertebrate diversity than in uncontaminated systems, although some trends of recovery are evident where effluents or emissions have been reduced (Gunn et al. 2010; Wesolek et al. 2010). By contrast, Spencer et al. (2008) found evidence of eutrophication and resultant increased invertebrate abundance and diversity downstream of metal mines.

2.4.3. Zooplankton and phytoplankton

Crustacean zooplankton species (e.g., *Daphnia* spp.) are sensitive test organisms that are commonly used in laboratory bioassays to assess the toxicity of mining effluents and metal concentrations in receiving waters (Kong et al. 1995) and therefore it is not surprising that field studies in metal-contaminated lakes showed significant adverse effects on zooplankton communities (Table 4). We found three long-term studies in metal-contaminated boreal lakes that showed that abundance and diversity can be significantly reduced among zooplankton communities, that these effects can persist for decades, and that signs of zooplankton recovery coincide with improving water quality conditions (Melville 1995; Yan et al. 2004; Valois et al. 2010). We reviewed five published studies on the impacts of mining contaminants on phytoplankton in boreal waters and the findings were similar to those for zooplankton communities (Table 4). Metal contamination in water bodies caused negative, but variable, effects on phytoplankton communities including reductions in diversity and shifts in community structure.

2.4.4. Mining – prognosis

Mining and processing of minerals and metals across the boreal zone can result in the release of effluents and emissions and can carry the risk of unintentional tailings pond seepage to natural waters. When toxic materials are present in effluents and emissions and are delivered to receiving waters at effective concentrations, they often have significant adverse and long-lasting effects on aquatic biodiversity. If mining development increases, or as more mines become abandoned, risks to aquatic biodiversity in receiving waters will increase accordingly and be incremental to the existing risks from current and abandoned mining operations (Ptacek et al. 2004).

These risks can be mitigated to some extent by careful mine management, effluent and emission controls, and improved technologies for tailings pond reclamation. The Mining Association of Canada, in conjunction with other nongovernmental and governmental bodies, is committed to the remediation of abandoned mines under the National Orphaned/Abandoned Mines Initiative (NAOMI 2009), but the task is daunting given the number of abandoned mines and the expense of remediation. There are more than 10 000 exploration and mining sites across Canada that require varying degrees of remediation (Tremblay and Hogan 2006), including at least 1300 abandoned mines in the boreal zone (CESD 2002; CDL 2005; Cowan et al. 2010). Natural treatment of mining effluents and acid mine drainage using constructed wetlands (Sobolewski 1996; Zhang et al. 2010) is a reclamation technology that would appear to be well suited for aquatic systems in boreal watersheds (given the prevalence of natural wetlands) but the practice does not seem to be widespread among mining sites across the boreal zone. Several long-term studies have demonstrated that water quality conditions in boreal watersheds can measurably improve with enhanced emission and effluent treatments and that these improved conditions can promote biological recovery (Keller and Yan 1991; Gunn et al. 1995; Keller et al. 1998, 2004, 2007). However, the recovery pathways are long and complex (Valois et al. 2010; Wesolek et al. 2010) and the barriers to recovery of aquatic biodiversity are not yet well understood (Yan et al. 2003, 2008). A better alternative to relying on remediation strategies to "clean up" after mining operations is to incorporate proactive environmental and conservation goals into the initial planning stages for further mining development on boreal watersheds (i.e., prevent or minimize damage in the first place). This process has begun in Ontario, for example, under the province's revised *Mining Act* and the Canadian Boreal Initiative (Wells et al. 2010).

The related issue of acidic deposition from airborne pollutants and their impacts on boreal water bodies is not resolved. Emissions of acidifying pollutants have been greatly reduced in the past couple of decades, by about 50% nationally with much higher reductions in some areas (CCME 2011). However, in other areas acid deposition still exceeds critical loads, acidifying emissions are still increasing, and biological recovery has been delayed (FPTC 2010). Coupled with persistent acid rain impacts is the emerging evidence of wide-scale calcium (Ca) depletion in forest watersheds that has been linked to severe reductions in some zooplankton (*Daphnia*) in lakes (Jeziorski et al. 2008, 2012). Accelerated Ca and other base cation leaching from soils after long-term acidic deposition has been shown to exceed natural replenishment processes, such as weathering, and to result in depletion of Ca stores in catchments (Watmough and Dillon 2003, 2004). This process of Ca loss from watershed soils also appears to be accelerated by forest harvesting (Watmough et al. 2005). Reduced Ca export to receiving waters has caused significant declines in lake water Ca concentrations, and these declines have caused a loss of sensitive crustacean species (Yan et al. 2008; Cairns and Yan 2009; Shapiera et al. 2012). The potential for cascading effects of lake water Ca declines was demonstrated by Korosi et al. (2012) who showed that elevated algal production was mediated through the effects of Ca decline on *Daphnia*. These impacts on aquatic biodiversity from the continuing threat of acidic deposition and from the serious Ca depletions in forest soils and lakes indicate increasing risks to food web interactions and stability in water bodies of the boreal zone. As mineral smelting and other industrial development increases, the likelihood of acidic deposition and other airborne pollutants with associated risks to aquatic biodiversity may also increase. Acknowledgement of these risks contributed to the impetus for the recent establishment of an environmental monitoring plan in western Canada to track airborne and waterborne pollutants (Environment Canada 2011).

2.5. Oil and gas exploration and development

Many of the risks to aquatic biodiversity for oil and gas exploration and development are similar to those identified for the mining of minerals and metals. The exploration corridors, access roads, pipelines, facility and well-drilling infrastructure, water extractions, water processing, tailings pond water and seepages, and emission discharges can all have impacts on natural hydrologic flow paths, water levels, water quality, aquatic and wetland habitats, and biodiversity. There are more than 220 000 active and inactive well sites drilled by the oil and gas industry in boreal Canada, mostly in the western boreal zone (Brandt et al. 2013), and this number is increasing at the rate of at least 10 000 new wells per year (Wells et al. 2010). There were over 1.5 million km of seismic lines by the mid 1990s for oil and gas exploration in Alberta, and the extent of these lines has increased by tens of thousands of kilometres each year since then (Machtans 2006). Activities associated with installation and maintenance of these exploration lines and other linear features can disrupt hydrological connectivity, and this loss of connectivity has implications for aquatic habitats and their biodiversity (Creed et al. 2011).

The most rapidly expanding, and arguably the most controversial, exploration and development related to oil and gas in the boreal zone is the mining of oil sands, primarily in the Athabascan region of northern Alberta where boreal wetlands are predominant on the landscape. Our search of the scientific literature on the impacts of oil and gas exploration and development on aquatic biodiversity over the past two decades was overwhelmed by publications pertaining to oil sands development. Conversely, the literature search for published studies from the boreal zone on aquatic impacts of conventional oil and gas exploration and development, including pipelines, returned very few results. We therefore focused our attention on oil sands development because it is a timely issue and is dominating the literature on impacts of oil and gas extractions.

Our review of oil sands impacts has relied primarily on peer-reviewed publications. A plethora of other reports and documents are available from industry and nongovernmental agencies from various web-based sources (e.g., Timoney 2010). It is difficult to produce a balanced overview of the risks and impacts of oil sands mining on aquatic biodiversity because the published and web-based information is polarized. Some commentators have indicated that there is a critical lack of reliable, available, and comprehensive environmental studies and information from active and burgeoning oil sands development (Timoney and Lee 2009; Gates 2010). Others contend that active and effective research and monitoring activities are providing relevant information to inform regulatory and assessment efforts (RAMP 2012; COSIA 2013).

Most of the current oil sands development is focused on surface mining and processing. Risks to aquatic biodiversity from surface mining arise from land and wetland clearing during site preparation, access road construction, diversion of natural drainage patterns, water extractions from source water areas, inadvertent leaks or spills from processing areas, seepage from tailings ponds, and deposition of airborne emissions (Kelly et al. 2010; Jordaan 2012). The effects of exploration and access roads on fish habitat, populations, and macroinvertebrates are similar to those described for forest management in Sect. 2.2.1 (e.g., Scrimgeour et al. 2008; and see Table 1) and are not reiterated here. Almost all of the recent published literature on the effects of oil sands development on aquatic biodiversity was focused on tailings ponds and their constituent water. Naphthenic acids released during bitumen extraction are the most toxic components of tailings pond water (Allen 2008). Seepages from bitumen processing and other aspects of oil sands development have been shown to elevate a suite of priority pollutant elements in the Athabasca watershed (Kelly et al. 2010) and may pose risk of toxic effects on aquatic organisms.

2.5.1. Fish

Of the seven studies we reviewed that have assessed the toxicity of oil sands tailings pond water to fish, all found significant adverse effects on various measurement end points at concentrations that occur in tailings ponds (Table 5). Survival, growth, and reproduction were all shown to be adversely affected by toxic compounds in tailings pond water. Reclamation efforts for tailings ponds and end pit lakes (one reclamation approach that essentially buries tailings and overlays them with water) will need to reduce concentrations of the toxic compounds in the tailings pond water if these ponds are to support fish populations. All studies assessed the effects of actual tailings pond water; none that we could find assessed the in situ effects of the tailings pond leachates in natural receiving waters. The deposition of toxic compounds from oil sands emissions could potentially present further risks to fish (Kelly et al. 2010), but the published literature did not test this possibility directly.

2.5.2. Macroinvertebrates

As early as 1979, a study was published that demonstrated the toxicity of oil sands tailings pond water to aquatic macroinvertebrates (Barton and Wallace 1979) and this was the harbinger of subsequent studies that showed generally adverse, although variable, effects on macroinvertebrates (Table 5). Wetlands receiving water from tailings ponds showed significant shifts in community structure with reductions in sensitive species and reciprocal increases in tolerant species (Bendell-Young et al. 2000). Given that oil sands development, with its associated emissions and tailings ponds, has been ongoing for over three decades, the number of published studies that have assessed the impacts on aquatic macroinvertebrates was surprisingly small (we found five).

2.5.3. Zooplankton, phytoplankton, and periphyton

A few published studies have assessed the toxicity of tailings pond water to plankton species, usually in laboratory bioassay settings. Similar to the findings for fish and macroinvertebrates, tailings pond water was found to be toxic to zooplankton (*Daphnia magna* as the test species) and algal groups (Table 5). Adverse effects on some phytoplankton species resulted in an altered community structure that favoured less sensitive species.

2.5.4. Macrophytes

The toxicity of oil sands tailings pond water and processing emissions to aquatic plants is particularly relevant because aquatic plants are common in the wetland-dominated landscape of the oil sands region and are thought to be potential phytoremediation and reclamation tools for tailings ponds and reclaimed wetlands (Trites and Bayley 2009). The studies we examined clearly demonstrated that this potential will require careful selection of tolerant and effective species, and an assessment of the ecological implications, because most aquatic plants were negatively affected by exposure to tailings pond water (Table 5). Armstrong et al. (2009) found that cattails (*Typha latifolia*), common reed grass (*Phragmites australis*), and hard stem bulrush (*Scirpus acutus*) were able to reduce the toxicity of naphthenic acids from tailings ponds to an aquatic crustacean by about 30%, although the growth of these plants was reduced by exposure to the tailings pond water.

2.5.5. Oil and gas development – prognosis

The risks of oil and gas development to aquatic biodiversity in boreal Canada are essentially a regional issue, with most of this industrial activity taking place in the western boreal zone. Nevertheless, within that region, oil sands development appears to pose significant risk of harm to aquatic biodiversity. The published literature currently available clearly demonstrates that toxic compounds in oil sands emissions and tailings ponds can be highly detrimental to an array of aquatic biodiversity if those toxic compounds reach natural receiving waters. To the extent that surface oil sands mining and bitumen processing are likely to increase in the future, risks to aquatic biodiversity will increase as risks of toxic compounds generated by oil sands development reaching receiving waters increase. It is less clear how often and in what concentrations those compounds occur in surrounding water

Table 5. Impacts of oil extraction from oil sands on aquatic biodiversity.

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
Fish	Oil sands tailings water	Alberta	Weeks	Fathead minnow spawning; sexual development	–; –	Kavanagh et al. 2011
	Oil sands tailings water	Alberta	Short term	Fathead minnow survival; brook stickleback survival	–; –	Bendell-Young et al. 2000
	Oil sands tailings water	Alberta	3 weeks	Gill; liver histopathology	–; –	Nero et al. 2006
	Oil sands tailings water	Alberta	12 days	Juvenile mortality; hatching; health; growth	–; –; –; –	Colavecchia et al. 2004
	Contaminated sediments	Alberta	Short term	White sucker egg survival; hatching; health; growth	–; –; –; –	Colavecchia et al. 2006
	Oil sands water	Alberta	28 days	Survival; hematology; gill histology	–; –; –	Farrell et al. 2004
	Oil sands reclamation ponds	Alberta	3 months and 10 months	Pathology; gill morphology	–; –	van den Heuvel et al. 2000
Macroinvertebrates	Oil sands tailings water	Alberta	4 weeks	Biomass; richness	–; –	Barton and Wallace 1979
	Oil sands tailings water	Alberta	Short term	Community structure; chironomid density	–; ++	Bendell-Young et al. 2000
	Oil sands tailings water	Alberta	1 year	Abundance and community structure	+ or - (species dependent)	Parsons et al. 2010*a*
	Oil sands tailings water	Alberta	1 year	Community structure	– or - (dependent on analysis)	Parsons et al. 2010*b*
	Oil sands reclaimed wetlands	Alberta	1 year	Abundance; biomass	–; -	Kovalenko et al. 2013
Zooplankton	Oil sands tailings water	Alberta	30 days	*Daphnia* survival	- (toxicity reduced in planted systems)	Armstrong et al. 2009
Phytoplankton and periphyton	Oil sands effluents	Alberta	Weeks	Green algal survival	–	Warith and Yong 1994
	Oil sands tailings water	Alberta	1 week	Phytoplankton community structure	- (shift to tolerant species)	Leung et al. 2001
	Oil sands tailings water	Alberta	Short term	Phytoplankton biomass; community composition	0; - (shift in composition)	Leung et al. 2003
Macrophytes	Oil sands tailings pond	Alberta	Weeks	Diversity; germination; growth	-; –; -	Crowe et al. 2002
	Oil sands effluent	Alberta	Weeks	Photosynthesis; growth	+; -	Bendell-Young et al. 2000
	Oil sands tailings water	Alberta	Weeks	Toxin uptake; survival	-; -	Armstrong et al. 2008
	Oil sands tailings water	Alberta	30 days	Growth; survival	–; – or – (nonionized)- or 0; 0 (ionized)	Armstrong et al. 2009
	Wetlands near oil sands	Alberta	Survey	Richness	-	Trites and Bayley 2009
	Oil sands tailings water	Alberta	Short term	Diversity; assemblages	–; –	Rooney and Bayley 2011

*The impact rating scale is as follows: –, large decrease; -, moderate decrease; 0, little or no measurable effect (where little is either a small or a brief change); +, moderate increase; and ++, large increase. End points are listed together, separated by a semicolon, for studies that examined multiple assessment end points.

bodies, and a recent monitoring program has been established to address this uncertainty (Environment Canada 2011).

Although a considerable volume of unpublished, usually web-based, information contends or infers impacts of oil sands development on aquatic biodiversity, the published literature is scarce on several outstanding issues. Intensified water quality monitoring and improved analytical technologies have demonstrated that a suite of compounds with known toxicities can be found in natural receiving waters of the oil sands region (Kelly et al. 2010), but empirical studies to determine their impacts on resident biota are few (Jordaan 2012). The development of engineering technologies and bioremediation methods for landscape and tailings pond reclamation is ongoing (Sawatsky et al. 1996), but comparatively little has been published on their effectiveness in reducing toxic compounds and in conserving or restoring aquatic biodiversity (e.g., Leung et al. 2001). Elevated concentrations of polycyclic aromatic hydrocarbons (PAHs) have been detected in lake sediments up to 90 km from oil sands development sites but the ecological consequences of increasing contaminant loads to regional lakes, coupled with climate-induced shifts in biotic community structure, are unclear and remain a critical knowledge gap (Kurek et al. 2013). With a few exceptions, most of the literature is focused on the toxicity of tailings ponds water while relatively few studies report on the effectiveness of reclamation strategies in conserving the aquatic biodiversity of wetlands and landscapes. Those that do, generally report incomplete ecological restoration after a decade or more of reclamation efforts (e.g., Bendell-Young et al. 2000; Trites and Bayley 2009; Rooney and Bayley 2012; Kovalenko et al. 2013). According to web-based information sources (e.g., RAMP 2012), these types of studies are underway but it appears that more public dissemination of research findings is needed. In addition, although most of the current research projects are focused on the impacts of surface mining and related development, below-ground (or in situ) oil extraction is poised to increase dramatically in the near future, potentially affecting an area 25 times larger than the current surface mining area (Schindler and Lee 2010). Because this is a relatively new oil extraction technique, the published literature contains little information on the potential impacts of in situ oil extraction activities on aquatic ecosystems and their biodiversity.

2.6. Peat mining

Peat is partly decayed, moisture-absorbing plant matter found in acidic, saturated soils of boreal wetlands. There are approximately 36.5 million ha of western peatlands extending from north-eastern Manitoba to north-eastern Alberta, and 12.5 million ha of eastern peatlands, mostly in the Hudson and James Bay Lowlands (Lemprière et al. 2013). In some areas, particularly Quebec, peat is commercially extracted for sale as a horticultural soil conditioner or as a biofuel. Peat extractions can pose a risk to aquatic biodiversity because the process generally requires the draining and ditching of wetlands, hydrological disruptions, and the disturbance of wetland overburden. In the context of the boreal zone, wetlands are the forest areas that are frequently or continuously saturated with water, including fens, bogs, marshes, swamps, and wooded pools, and any of those containing >40 cm depth of peat development can be classified as peatlands (Webster et al. Manuscript in preparation). Disturbance of boreal wetlands is of particular concern because boreal wetlands are unique ecotypes, are prevalent in boreal forest landscapes, and are integral contributors to overall boreal ecosystem services, such as production and storage of clean water, provision of critical habitats for several species, and sequestration of carbon (Wieder and Vitt 2006; Tarnocai et al. 2009; Nyman 2011; Wells et al. 2010).

Peat mining is not the only type of natural resource extraction that threatens the integrity of wetlands. The most ominous threat to the biodiversity associated with boreal wetlands is the loss of wetland habitat from conversion to other land uses. Although most wetland losses are occurring in southern Canada as a result of agricultural and urban development, approximately 10 000 km^2 of boreal wetlands (primarily peatlands) have been drained or disturbed for natural resource development over the last four decades. About 90% of the 10 000 km^2 was destroyed by flooding for hydroelectric impoundments, while the remaining 10% was lost from drainage for forest harvesting, peat extractions, oil sands mining, and conversion to agriculture (Poulin et al. 2004; Foote and Krogman 2006; FPTC 2010).

Peat mining is the only type of natural resource development that is specifically targeted at peatland forms of wetlands and is therefore considered separately here. Wetlands in general can be prolific sources of aquatic biodiversity among all the biotic groups we use as bioindicators, but the literature on the impacts of boreal peat mining on aquatic biodiversity is limited and is mostly focused on semiaquatic or bog plants (Table 6). Only one study was found that assessed the effects of peat drainage on fish and it was inconclusive, although there was some indication of reduced abundance (Clement et al. 2009). We found five studies that assessed the effects on amphibians and aquatic invertebrates, and all reported adverse effects to varying degrees (Table 6). Of the 10 studies we reviewed that reported effects on vegetation (semiaquatic vegetation or as part of total ground vegetation community), almost all reported adverse effects in terms of reductions in the number and diversity of bog species and shifts in community structure toward non-bog species (Table 6).

2.6.1. Peat mining – prognosis

With increasing interest in using peat as a biofuel (Telford 2009), peat extractions from boreal wetlands could increase. However, at present, risks to aquatic biodiversity from peat mining are not a significant issue across the entire boreal zone in Canada, or even on a regional basis. Less than 0.02% of Canada's total peatlands are under threat from peat mining (Daigle and Gautreau-Daigle 2001). Nevertheless, at a local scale, peat mining clearly has significant harmful effects on bog plant biodiversity with potential for secondary adverse impacts on wetland habitats and aquatic biodiversity. These impacts can be mitigated or managed through the preservation of ecologically unique wetlands, effective restoration or reclamation efforts, and increased research into environmentally sustainable strategies and practices (Daigle and Gautreau-Daigle 2001; Chapman et al. 2003). These environmental impact considerations are key to discussions concerning the potential use of peat as a bioenergy source.

3. Summary and synthesis

The natural resource development activities with the largest spatial extent of risk to aquatic biodiversity across the boreal zone are associated with forestry and mining. The forest sector poses potential risks to aquatic biodiversity through changes to the composition and structure of watershed and riparian forests, through contamination of water bodies by forest pesticides, and through effects of effluents from forest product manufacturing. Considerable advances have recently been made in mitigating the impacts of these disturbances on aquatic biodiversity as recent studies generally show little or short-term impacts. The prognosis overall for the forest sector's impacts on water resources and aquatic biodiversity is improving with advanced technologies in pulp mill effluent treatment, and ecosystem-based forest management regulations and practices. Some uncertainties remain about the impacts of forest management, including the frequency and extent of spatially and temporally cumulative effects, the potentially new risks associated with forest biomass harvesting for biofuels, and the effectiveness of natural disturbance pattern emulation as a means to sustain forest and aquatic biodiversity.

Table 6. Impacts of peat mining on aquatic biodiversity.

Indicator	Disturbance type	Location	Length of study	Assessment end point	Impact*	Reference
Fish	Peat mining	New Brunswick	3 years	Abundance	- or 0 (inconclusive)	Clement et al. 2009
Amphibians	Peat mining	New Brunswick	Survey	Frog movement	-	Mazerolle 2001
	Peat mining	Quebec	Survey	Richness; abundance	-; -	Mazerolle 2003
	Peat mining	Quebec	Survey	Frog abundance; reproduction	-; –	Mazerolle and Cormier 2003
Macroinvertebrates	Peatland drainage	New Brunswick	2 years	Implied impacts from increased sediments	-	St-Hilaire et al. 2006
	Peatland drainage	Quebec	Survey	Diptera richness	-	Taillefer and Wheeler 2010
Macrophytes	Peatland drainage	Europe	Review	Bog or semi-aquatic species	–	Laine et al. 1995
	Peatland clear-cutting	Manitoba	Survey	Total plant diversity; bryophyte and lichen diversity; cover	+; -; -	Locky and Bayley 2007
	Peatland clear-cutting and extraction	Quebec, not boreal	Survey	Species richness	–	Tousignant et al. 2010
	Peat mining	Quebec, not boreal	8 years	Community composition; structure	–; –	Berube and Lavoie 2000
	Peat mining	Quebec	Survey	Community composition	–	Lavoie and Saint-Louis 1999
	Peat mining	Quebec	Survey	Community composition	–	Robert et al. 1999
	Peat mining	Quebec	Survey	Diversity; abundance; composition	-; –; –	Poulin et al. 1999
	Peat mining	Quebec	Survey	Mosses; herbs; shrubs	–; –; 0 (more edge species)	Campbell et al. 2003
	Peat mining	Quebec	Survey	Colonization of vascular species; sphagnum	0; –	Girard et al. 2002
	Peat mining	Quebec and New Brunswick	Survey of recovery	Bog species cover	– or -	Poulin et al. 2005
	Peat mining	Sweden	Short term	Richness; abundance	-; - or 0 (species dependent)	Soro et al. 1999

*The impact rating scale is as follows: –, large decrease; -, moderate decrease; 0, little or no measurable effect (where little is either a small or a brief change); +, moderate increase; and ++, large increase. End points are listed together, separated by a semicolon, for studies that examined multiple assessment end points.

Mitigation of the effects of mining on aquatic biodiversity can be more problematic. The inevitable mining by-products contain many toxic elements that can discharge from active mining and processing facilities and leach from historic mines and tailings ponds. There are indications that new mines will be developed further into the northern parts of the boreal zone, and this coupled with the large number of abandoned mines exacerbates these risks to aquatic biodiversity as many older sites continue to leach toxic compounds into boreal watersheds. However, the mining sector recognizes and is responding to these risks by improved mining management, and effluent and emission control technologies (e.g., Natural Resources Canada's Green Mining Initiative www.nrcan.gc.ca/minerals-metals/technology/4473, and the Clean Mining Alliance www.miningandexploration.ca/sustainability/article/new_clean_mining_association_is_formed/). The effectiveness of these newer technologies for protecting aquatic biodiversity cannot yet be determined from the published literature, and the recent national assessment of mining effluent impacts demonstrated that significant adverse (often inhibitory) effects on the key indicator organism (fish) are still detected at many sites (Environment Canada 2012). Similarly, controls on metal smelter emissions have improved water quality in the down-wind footprint of smelting operations, but biological recovery in those water bodies is slow, complex, and incomplete.

River regulation and impoundment for hydroelectric power generation always have large, measurable impacts on wetland and aquatic biodiversity in flooded areas and in downstream reaches. Insofar as the number of hydroelectric installations is expected to increase across the boreal zone, the risks to wetland and aquatic ecosystems from this sector are also expected to increase. Mitigation options are limited, but discharge regulations to mimic more natural flow regimes, the movement toward smaller, more efficient power stations, and the careful selection and placement of new installations in boreal watersheds may reduce overall impacts on aquatic and wetland ecosystems. One approach to limiting the impacts of hydroelectric development on aquatic biodiversity is to increase reliance on "greener" technologies to produce power, although many of those technologies are not without social and environmental issues of their own. An integrated, comprehensive strategy for power production and consumption that included reducing overall electricity demand, putting generation sources as close as possible to major demand centres, and congregating power generation in already disturbed environments rather than continually encroaching on undeveloped boreal watersheds could contribute to overall reduction in risk to biodiversity.

Another natural resource sector encroaching on boreal wetlands and watersheds is the growing oil sands industry. Oil sands mining and processing affect a relatively small portion of the total boreal zone but have the potential to have significant adverse effects on local or regional wetlands, aquatic biodiversity, and overall aquatic ecosystem health at that spatial scale. Oil sands tailings pond water, seepages, and emissions have been shown to contain materials at concentrations that are toxic to most aquatic organisms, and as this sector's activity increases the frequency of tailings ponds, potential seepages, and contaminants from emission releases could increase accordingly. Surface mining activities cause obvious disturbances to wetlands, hydrological flow paths, and boreal landscapes, whereas the impacts of below-surface extractions are largely unknown. Research and monitoring efforts to assess, predict, and mitigate these impacts have increased, and advances are being made in cleaner technologies and in enhanced reclamation strategies. The published literature has little information on the effectiveness of these newer technologies, and many uncertainties remain and continue to provide challenges to conserving and restoring functional wetlands and aquatic ecosystems in the oil sands region.

Peat mining affects only a small portion of the Canadian boreal zone, but at a local scale it can have significant harmful effects on bog plant biodiversity with potential for secondary adverse effects on wetland habitats and aquatic biodiversity. Although the spatial extent of risk to aquatic biodiversity is currently small, increasing interest in peat as a biofuel could contribute to the overall loss of wetland habitats, if peat harvesting for biofuel increases. From the few studies we found, it was apparent that mitigation measures for peat mining are few but that current emphasis is on enhancing restoration and reclamation technologies for peat bogs, and these could offset risks to biodiversity from peat mining in the longer term.

For most of the disturbance types related to natural resource development, a current or recent status assessment was difficult. Despite the prevalence of natural resource development in Canada's largest forest ecosystem, the published literature on the status of and trends in various indicators of aquatic biodiversity across many of the disturbance types was limited. The greatest amount of published information relevant to the boreal zone that was available for assessing risks and impacts was for forest harvesting and roads, followed by pulp and paper effluents, hydroelectric impoundments, current-use forest pesticides, metal mining, oil sands development, and peat mining. Many of the published assessments relied on small-scale experimental or manipulative studies rather than field-based measurements or monitoring. Difficulties in assessing recent status and trends extend to uncertainties associated with forecasting future status and trends. Across most of the boreal zone, there is a lack of coordinated, consistent monitoring programs or data collection for many of the bioindicators and disturbance types discussed in this review.

Climate change will confound and potentially intensify the impacts of natural resource development on aquatic biodiversity. Natural resource managers and regulators will face increasing challenges in ensuring the conservation of healthy aquatic ecosystems and their biodiversity as they deal with uncertainties associated with the severity, timing, frequency, and interactions of climate change impacts (Schindler 1998*b*, 2001; Keller 2007; Heino et al. 2009). Climate change can affect aquatic biodiversity directly through warming water temperatures, altered hydrological regimes and flow patterns, and shifts in seasonal timing of temperature and flow maximums, as well as indirectly through increased severity and frequency of forest fires, storm events, and pest insect outbreaks, changes to watershed vegetation composition, warming soils with alterations in watershed biogeochemical processes, and the export of waterborne nutrients to receiving waters (Keller 2007). Increased research effort and systematic, long-term monitoring will be required to further our understanding of the confounding effects of climate change on the impacts of natural resource development and how to mitigate or adapt to them.

In conjunction with climate change influences, other disturbances will increasingly interact with and confound the impacts of natural resource development on aquatic ecosystems. This will complicate management and mitigation strategies. There is increasing awareness and concern that the interacting and cumulative effects of several concurrent stressors on aquatic ecosystems are poorly understood but can have profound influences on the sensitivity and resilience of aquatic communities with respect to any individual stressor (Lowell et al. 2000; Scrimgeour et al. 2008; Keller 2009; Desellas et al. 2011; Seitz et al. 2011; Dube et al. 2013). Stressors that can potentially interact with climate change and resource development disturbances to affect boreal watersheds and their aquatic biodiversity include continuing acidification, other airborne pollutants, calcium declines, invasive aquatic and terrestrial species, urban expansion and municipal effluents, agricultural water consumption and nutrient inputs, ecotourism, and recreational activities. Difficulties arise from identifying and quantifying causality among multiple stressors, from addressing time lags between disturbance or stressor events and biological responses, from varying and overlapping recovery times, and from complex linkages among interacting effects and their responses. The cumulative effects of these stressors coupled with resource development disturbances on boreal watersheds remain largely unknown. More importantly, the ecological thresholds for these cumulative effects (that is, the point at which aquatic ecosystems and their biodiversity cannot recover to a desired state within a reasonable time frame) are also unknown and remain gaps in our knowledge about risks to aquatic biodiversity.

4. Conclusion

The literature indicates that there are several natural resource development activities that pose a risk of harm to aquatic biodiversity through alterations of watershed features, changes in aquatic and wetland habitats, and contamination of water bodies from sedimentation, effluents, toxic seepages, and emissions. To the extent that natural resource development increases and expands into previously undeveloped areas of the boreal zone, these risks to aquatic biodiversity could increase accordingly. Any new threats arising from expanded resource development would be incremental to some persistent impacts from legacy resource development operations, such as historic and abandoned mines, and continuing emission-related acidification. New and improving technologies and regulations for many resource development activities have potential to mitigate or offset risks to aquatic biodiversity, but an assessment of the effectiveness of these newer technologies for the protection of aquatic biodiversity is not yet available in the published literature.

Recent environmental initiatives suggest that when the use of "greener" technologies and regulations for resource development are combined with practical conservation planning and increased stewardship in watershed management, risks to aquatic ecosystem integrity can be reduced. The Canadian Boreal Forest Agreement among multiple stakeholders and interest groups is an example of a broad planning framework within which the conservation of biodiversity plays a prominent role; the agreement sets the stage for achieving the highest standards of sustainable resource development (Wells et al. 2010). A watershed or regional-level conservation planning approach that explicitly includes the protection of water and other ecosystem services would greatly improve the assessment and management of water resources and move us toward sustaining healthy aquatic ecosystems across Canada's boreal zone (Schindler and Lee 2010). Furthermore, a national, coordinated, multistakeholder program of monitoring and research focused on adaptive resource development and management would increase our capacity to assess risks and mitigate effects on aquatic ecosystems of the boreal zone.

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